Modelling the Multispecies Fishery of Chwaka Bay, Zanzibar – Basis for Exploration of Use and Conservation Scenarios

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The imagination of nature is far, far greater than the imagination of man.

I stand at the seashore, alone, and start to think. There are the rushing waves
. . . mountains of molecules each stupidly minding its own business
. . . trillions apart
. . . yet forming white surf in unison.

Ages and ages
. . . before any eyes could see
. . . year after year
. . . thunderously pounding the shore as now.
For whom, for what?
. . . on a dead planet, with no life to entertain.

Never at rest . . . tortured by energy
. . . wasted prodigiously by the sun
. . . poured into space.
A mite makes the sea roar.

Deep in the sea, all molecules repeat the patterns
of one another till complex new ones are formed.
They make others like themselves
. . . and a new dance starts.

Growing in size and complexity
. . . living things, masses of atoms, DNA, protein
. . . dancing a pattern ever more intricate.
Out of the cradle onto the dry land
. . . here it is standing
. . . atoms with consciousness
. . . matter with curiosity.

Stands at the sea
. . .wonders at wondering
. . . I. . . a universe of atoms
. . . an atom in the universe.

Richard P. Feynman, The Value of Science, 1955

Abstract

Small-scale fisheries are essential livelihood providers for coastal communities in developing countries and are often their prime protein supplier. Particularly in the Western Indian Ocean, the dependency on small-scale fisheries is very high. Perceived decreases in individual catches and the use of destructive gears and small mesh sizes has led to concerns for an unsustainable use of fisheries resources in many coastal areas throughout the Western Indian Ocean. This, together with the importance of small-scale fisheries for food-security in the region, makes their management of primary importance. However, the lack of knowledge on the state and dynamic of fisheries strongly impedes the potential for the development of proper management plans. This situation is very prevalent in the semi-autonomous island state Zanzibar (Tanzania) and, in particular, in Chwaka Bay, located on the east coast of the island. The bay's resources are believed to show serious signs of overfishing. Particularly the increasing use of the destructive dragnets has led to resource concerns and to strong conflicts between fishermen. Despite several management attempts, this situation has not changed over the last two decades.

The aim of the present dissertation is, therefore, to assess the status of Chwaka Bay's ecosystem and its fishery, as well as to evaluate different potential use scenarios for sustainable fisheries management. Furthermore, using Chwaka Bay as a reference site, the dissertation aims at approaching the answer to the question of the sustainability of Zanzibar's nearshore fisheries.

The approach used in this dissertation is twofold: 1) length-based stock assessments were conducted for six of the main target species (i.e. *Siganus sutor*, *Lethrinus lentjan*, *Lethrinus borbonicus*, *Lutjanus fulviflamma*, *Leptoscarus vaigiensis* and *Scarus ghobban*) to evaluate their current exploitation status in relation to safe biological limits and to estimate fishing mortality as well as biomass across the different length-classes; and 2) a trophic flow model of the bay was constructed using *Ecopath with Ecosim/Ecospace* to describe the current state and flow structure of the system to evaluate overall and gear-specific fishing impacts on the ecosystem and the fishing community and to simulate potential management scenarios. The data used in both approaches were collected through an extensive field survey conducted over an annual cycle in 2014. These data include 1) information on gear-specific catch composition, catch weight and fishing effort; 2) the length frequency distributions of the six key species; and 3) cost-related information (e.g. fuel and gear costs).

A review of the literature about the state of Zanzibar's fisheries reveals that no fisheries assessments have been conducted after 2000. Analyses of the annual reported landings between 1990 and 2014 suggest that, except for clupeioids, none of the target groups of the fishery can be classified as overfished. Most studies aimed at evaluating the status of Zanzibar's resources have been focusing on ecological surveys and fishermen's perception. None of the ecological surveys appropriately link fishing effort and/or fishing pattern with resource state nor provide tangible thresholds for management.

The stock assessment of the key target species of Chwaka Bay suggests that the exploitation rate of three out of the six target species (i.e. *Siganus sutor, Lethrinus borbonicus, Lethrinus lentjan*) exceeds recommended levels ($E_{0.1}$). Despite high juvenile retention rates and length at first capture being below optimum length at first capture, fishing mortalities are highest for adults. Due to the nursery characteristics and the topography of the bay, juveniles might occur in higher abundances and larger fish may concentrate further outside the bay area. Consequently, an increase in mesh size only seems economically viable, if the radius of the fishery was increased to capture larger specimens outside the shallow bay area.

The trophic model indicates that the Chwaka Bay ecosystem is comparatively mature, with relatively high transfer efficiencies. The bay is strongly bottom-up driven, with biomass concentrations around the first and second trophic level and a low overall fish biomass. The strongest impact on the ecosystem is exerted by dragnets and traps. Both gears potentially destabilize the ecosystem by reducing the biomass of top-down controlling key species. Together with handlines, dragnets and traps are the least selective fishing methods. In addition, traps exert the highest fishing pressure on 4 out of the 6 selected key species. While the dragnet fishery is the least profitable, it also provides the highest number of jobs for the fishing community, as it is a labor-intensive fishing method. In contrast, longlines and gillnets are more selective and more profitable.

Simulations of different use scenarios suggest that the elimination of dragnets would lead to the highest increase in overall fish biomass and profits. Nevertheless, this scenario would leave 58 % of fishermen without job and is, therefore, not feasible under the current lack of alternative livelihoods and the high dependency on fisheries resources. The complete reallocation of dragnets is likewise not feasible, since current effort is already high, and a further increase will lead to strong biomass reductions of target species and losses in individual profits of fishermen. Without compromising individual profits (-20 %) and biomass structure of the ecosystem (-30 %), the effort of the main gears (i.e. traps, dragnets and handlines) do not allow for a large increase (1.2 - 1.4 fold), while a higher increase is possible for floatnets, gillnets and longlines (3.4 to 4.2 fold). However, this scenario would still leave 37 % of fisher without jobs.

In conclusion, the fishery of Chwaka Bay is fully exploited with some groups experiencing overfishing and does not provide scope for further expansion. Emperor fish (Lethrinidae) together with similar vulnerable target resources such as Serranidae and Mullidae might be unsustainably harvested throughout the island. However, the concern of a general overexploitation of Zanzibar's resources could not be confirmed. Furthermore, this dissertation challenges two common beliefs: 1) that high amounts of undersized fish in the catches is equivalent of unsustainable fishing; and 2) that the use of illegal gears and the decrease in individual catches are appropriate indicators for the state of fisheries resources. These findings, along with the general low level of information on fisheries resources in the Western Indian Ocean, highlight the need for proper fisheries assessments to support decision-making for sustainable management. In addition, the findings of the present dissertation suggest that neither a total ban of dragnets nor a complete reallocation of dragnet fishermen is currently feasible. The lack of recognition for the capacity of dragnet boats to absorb surplus labour and the marginalization of this group of fishermen is likely hindering the development of feasible management plans aimed at regulating their use. In order to stop the use of dragnets on Zanzibar, fisheries management should focus on initializing an effort control of this gear, while simultaneously investing in the diversification of livelihoods.

Zusammenfassung

Kleinfischereien stellen für die einheimische Bevölkerung vieler Küstengebiete weltweit die primäre Lebensgrundlage sowie eine wichtige Nahrungsquelle dar. Vor allem in den Küstengemeinschaften des Westindischen Ozeans ist die Abhängigkeit von solchen Kleinfischereien sehr hoch. Sinkende Fangraten, Anlandungen großer Mengen untermaßiger Fische und die Nutzung von destruktiven Fanggeräten haben hier zu einer wachsenden Sorge über die Zukunft der Küstenökosysteme geführt. Dies zusammen mit der Notwendigkeit der Kleinfischerei für die Nahrungssicherheit der Bevölkerung verleiht der nachhaltigen Bewirtschaftung der Fischbestände eine entscheidende Bedeutung. Mangelnde Kenntnisse über die Lage der Bestände und die Dynamik der Fischerei beeinträchtigen jedoch stark die Entwicklung geeigneter Managementpläne. Besonders ausgeprägt ist diese Situation in dem halbautonomen Inselstaat Sansibar, vor allem in Chwaka Bay an der Ostküste der Insel. Die Fischbestände der Bucht gelten seit vielen Dekaden als nicht nachhaltig befischt. Vor allem die verstärkte Nutzung von illegalen Schleppnetzen hat zu einer großen Sorge um die Küstenressourcen sowie zu starken Konflikten unter den Fischern geführt. Trotz verschiedener Managementversuche hat sich die Situation in den letzten zwei Jahrzehnten nicht verbessert.

Die vorliegende Studie hat daher als übergreifendes Ziel den Zustand der Fischerei und des Ökosystems in Chwaka Bay zu bestimmen sowie potenzielle nachhaltige Nutzungsszenarien zu untersuchen. Darüber hinaus soll anhand des Beispiels von Chwaka Bay der Zustand der Küstenfischereien Sansibars bewertet werden.

Hierzu verwendet die vorliegende Studie zwei Ansätze: 1) Mithilfe von längenbasierten Methoden zur Bestandsabschätzung wurden die Ausbeutungsraten von sechs Hauptfangarten (*Siganus sutor*, *Lethrinus lentjan*, *Lethrinus borbonicus*, *Lutjanus fulviflamma*, *Leptoscarus vaigiensis* und *Scarus ghobban*) im Verhältnis zu ihren nachhaltigen biologischen Grenzen ermittelt und die Bestandsgröße sowie fischereiliche Sterblichkeit über die unterschiedlichen Längenklassen berechnet; 2) Mithilfe von *Ecopath with Ecosim/Ecospace* wurde ein Nahrungsnetzmodel erstellt, um einerseits die Struktur der trophischen Flüsse zu beschreiben und andererseits die unterschiedlichen Einflüsse der verschiedenen Fangmethoden auf das Ökosystem und die Fischergemeinschaft zu untersuchen.

Die Grundlage der Analyse bildet eine im Verlauf des Jahres 2014 durchgeführte umfangreiche Felduntersuchung. . Die dabei gesammelten Daten beinhalten: 1) die Artenzusammensetzung des Fanges, Fangmenge und Fischereiaufwand der angewandten Fangmethoden; 2) Längenfrequenzen der sechs Hauptfangarten in den

Fängen; 3) kostenbezogene Informationen (z.B. Kraftstoffkosten und Kosten der Fanggeräte).

Die Literaturdurchsicht hat ergeben, dass seit dem Jahr 2000 keine Studie zur Beurteilung der Fischerei auf Sansibar mehr erfolgt ist. Offizielle Anlandungsdaten zeigen, dass keine der Zielgruppen, mit Ausnahme der Gruppe clupeioids, als überfischt eingestuft werden kann. Des Weiteren hat sich herausgestellt, dass ein Großteil der zum Thema Befischungsstatus durchgeführten Studien, ökologische Bestandsaufnahmen und Befragungen der lokalen Fischer darstellt. Keine dieser Studien hat die Ressourcen in Zusammenhang mit dem Fischereiaufwand und –muster gesetzt, geschweige denn greifbare Grenzwerte für ein nachhaltiges Management geliefert.

Die vorliegenden Analysen zum Bestandszustand der Hauptfangarten von Chwaka Bay haben ergeben, dass drei der untersuchten Arten eine Befischung außerhalb sicherer biologischer Grenzen aufweisen. Allerdings zeigte sich auch, dass trotz hoher Anlandungen untermaßiger Fische, der Fischereidruck auf adulte Individuen am größten ist. Aufgrund der Wassertiefe und der in der Bucht vorkommenden Mangroven und Seegraswiesen, die für viele Fische als Aufzuchts- und Nahrungsgebiet dienen, sowie der geringen Wassertiefe ist es fragwürdig, ob eine Erhöhung der Maschenweiten für die Fischer profitabel wäre.

Das trophische Model hat gezeigt, dass Chwaka Bay ein vergleichbar reifes, produktives Flachwassersystem darstellt, das von Primärproduzenten und Invertebraten dominiert wird. Auffallend ist jedoch die geringe Fischbiomasse im System. Die Reusen- und Schleppnetzfischerei haben den stärksten Einfluss auf das Ökosystem sowie auch auf die Fänge der anderen Fischereien. Beide Fangmethoden haben eine potenziell destabilisierende Wirkung auf das Ökosystem, indem sie die Biomasse solcher Arten reduzieren, die eine wichtige top-down Kontrollfunktion übernehmen. Zusammen mit der Handleinenfischerei sind die Schleppnetz- und Reusenfischerei am wenigsten selektiv. Zudem übt die Reusenfischerei den stärksten Fischereidruck auf vier der sechs Hauptfangarten aus. Obwohl die Schleppnetzfischerei am wenigsten gewinnbringend ist, beschäftigt sie über die Hälfte aller Fischer und stellt somit eine wichtige Institution für die lokale Bevölkerung dar. Im Vergleich zu den zuvor genannten Fangmethoden ist die Benutzung von Langleinen und Kiemennetzen selektiver und lukrativer.

Die Ergebnisse der Simulierung von unterschiedlichen Nutzungsszenarien deuten an, dass eine Abschaffung der Schleppnetzfischerei zu einer starken Erhöhung der Fischbiomasse sowie der Profitsteigerung aller Fischereien führen würde. Da eine derartige Maßnahme allerdings auch einen Verlust von 58 Prozent der Arbeitsplätze in der Fischerei bedeuten würde, ist sie unter dem gegebenen Mangel an alternativen

Einkommensmöglichkeiten nicht umsetzbar. Eine vollständige Umverteilung der Schleppnetzfischer auf andere Fangmethoden hingegen würde zu einer starken Abnahme der Biomasse der befischten Gruppen im System sowie zu einer starken Verringerung der fischereilichen Gewinne führen. Aufgrund des bereits sehr geringen Einkommens der Fischer stellt somit auch dieses Szenarium keine sinnvolle Lösung dar. Ohne die individuellen Profite der Fischer (-20 Prozent) und die Biomassestruktur des Ökosystems (-30 Prozent) stark zu beeinträchtigen, kann der Fischereiaufwand der Hauptfangmethoden (Reusen-, Schleppnetz- und Handleinenfischerei) nur wenig (1.2- bis 1.4-fache) und der Kiemen-, Langleinen- und Umschließungsnetzfischerei mäßig (3.4- bis 4.2-fache) erhöht werden. Dies bedeutet immer noch einen Verlust von 37 Prozent der Arbeitsplätze in der Fischerei.

Zusammenfassend weisen die Ergebnisse daraufhin, dass die Fischerei in Chwaka Bay voll genutzt und manche Gruppen des Systems überfischt werden. Folglich besteht keine Expansionsmöglichkeit der Fischerei. Zudem ist es wahrscheinlich, dass Lethrinidae und andere anfällige Fischfamilien, wie zum Beispiel Serranidae und Mullidae auf der gesamten Insel überfischt werden. Nichtsdestotrotz konnten die Bedenken hinsichtlich einer grundsätzlichen Überfischung der Fischereiressourcen Chwaka Bays nicht bestätigt werden. Demgegenüber stellt die vorliegende Dissertation zwei gängige Überzeugungen in Frage: 1) Hohe Anlandungen untermaßiger Fische sind gleichzusetzen mit nicht nachhaltigen Fischereipraktiken; 2) die Nutzung illegaler Fangmethoden und die Abnahme individueller Fangraten sind geeignete Indikatoren zur Einschätzung des Fischereizustandes. Betrachtet man dies im Zusammenhang mit dem grundsätzlichen Mangel an Informationen über Fischereiressourcen im Westindischen Ozean, wird deutlich, dass fischereiliche Studien unabdingbar sind für eine angemessene Einschätzung der Fischerei. Letztendlich zeigt die vorliegende Dissertation deutlich, dass ein vollständiges Verbot der Schleppnetzfischerei nicht umsetzbar ist. Die mangelnde Anerkennung der Aufnahmefähigkeit von überschüssigen Arbeitskräften der Schleppnetzfischerei und deren Ausgrenzung erschwert die Entwicklung von umsetzbaren Plänen zur Einschränkung ihres Fischereiaufwandes. Um eine nachhaltige Befischung des Ökosystems zu ermöglichen, ist zunächst die Einführung einer Aufwandsbeschränkung zu empfehlen, begleitet von Plänen zur Diversifizierung der Einkommensmöglichkeiten.

Ikithiri

Uvuvi mdogo au wa kienyeji ni muhimu kwa maisha ya jamii za nchi zinazoendelea na ni chanzo kikuu cha chakula aina ya protini. Nchi za magharibi ya hindi ndio hasa wanaotegemea uvuvi wa aina hii. Kutokana na kupungua kwa wingi wa samaki na kuongezeka kwa matumizi ya zana za kuvulia haramu au zisizokubalika na nyavu zenye macho madogo imeleta mtafaruku katika maeneo mengi ya ukanda wa magharibi ya hindi. Ukizingatia ya kuwa uvuvi mdogo ni muhimu kama chanzo cha chakula inabidi kuzingatia umuhimu wa kuwepo kwake, kuudhibiti na kuulinda uvuvi wa aina hii. Lakini kutokana na upungufu wa taarifa kuhusu hali halisi ya uvuvi inaletea ugumu wa kuwa na nyenzo nzuri za udhibiti wa uvuvi huu. Hali hii pia inaikabili Zanzibar (Tanzania) na hasa ghuba ya Chwaka ambayo ipo mashariki ya kisiwa cha Unguja. Inasaidikika ya kuwa kuna upungufu mkubwa wa samaki katika ghuba ya Chwaka. Hasa kutokana na uongezekaji wa matumizi ya nyavu za kukokota na imeleta mgongano wa wavuvi. Ingawa kumefanywa juhudi nyingi kuondosha jambo hili lakini bado halijabadilika kwa zaidi ya karne mbili sasa zilizopita.

Dhamira ya utafiti huu ilikuwa kuangalia hali halisi ya mazingira ya bahari ya Chwaka na uvuvi uliopo na pia kuangalia uwezo wa kuanzisha ulinzi na dhibiti endelevu wa uvuvi. Ingawa Chwaka imetumika kama eneo la utafiti lakini taarifa zitakazopatikana zinaweza kutumika katika maeno mengine pia.

Njia za utafiti zilizotumika zilikuwa kama ifuatavyo:1) kutumia ukubwa wa samaki wa aina sita ambao ni hawa (i.e. *Lethrinus lentjan*, *Lethrinus borbonicus*, *Lutjanus fulviflamma*, *Leptoscarus vaigiensis* na pono) kuangalia wanavyovuliwa kwa njia ya usalama na wanavyovuliwa katika ukubwa mbalimbali ili kupata uelewa wa wingi wa aina hii ya samaki; na 2) Modeli ya uwiano wa mfumo wa vyakula vinavyoliwa na samaki ulitaarishwa kwa kutumia modeli inayojulikana kitaalamu kama *Ecopath na Ecosim/Ecospace* kuelezea hali halisi ya namna ya mfumo wa chakula unavyoliwa na samaki ulivyo hivi sasa ili kuelewa hali halisi ilivyo pamoja na namna uvuvi unavyo athiri mazingira na jamii na kutathmini mbinu mbali mbali ambazo zinaweza kutumika kudhibiti uvuvi. Taarifa zilizotumika zimepatikana wakati wa kufanya utafiti kwa muda wa mwaka mmoja katika mwaka wa 2014. Taarifa zilizotumika ni 1) taarifa za kipato kwa zana zilizokusudiwa, aina, uzito na nguvu ya kuvua; 2) Urefu wa aina 6 za samaki waliovuliwa na 3) gharama za kuwapata hawa samaki (e.g. bei ya mafuta, zana nk) ukiangalia taarifa za uvuvi ziliopo. Hakujafanyika utafiti wa kuchunguza wingi wa samaki waliopo baharini tokea mwaka 2000. Mchanganuo wa samaki wanaovuliwa tokea mwaka 1990 hadi 2014 unaonyesha ya kuwa ukiwaacha samaki aina ya dagaa samaki wote wengineo huwezi kusema wamevuliwa kupindukia. Tafiti nyingi zinaangalia mazingira zaidi na nini wavuvi wanasema. Lakini hata hivyo hizo taarifa za mazingira hazihusishi nguvu ya kuvua, msimu na haitoi ushauri ya mbinu bora za ulinzi na udhibiti.

Kuangalia wingi wa samaki muhimu wanaovuliwa Ghuba ya Chwaka inaonyesha mavuvi ya aina tatu ya samaki yamepitiliza kiwango cha uvuvi unaokubalika (i.e. *Siganus sutor, Lethrinus borbonicus, Lethrinus lentjan*) ($E_{0.1}$). Ingawa kuwepo kwa kuvuliwa samaki wadogo na wenye urefu mdogo ukilinganisha na ule urefu wa kuvuliwa unaokubalika. Wengi wa samaki wanovuliwa ni wale ambao wamefikia urefu wao wa mwanzo wa kukuwa (yaani tayari kuzaa). Kutokana na hali ya Ghuba ya Chwaka kuwa ni eneo la mazalio ya samaki, samaki wachanga huwa wengi na samaki wakubwa hukimbilia nje ya ghuba. Kwa hivyo ongezeko la ukubwa wa macho ya nyavu inonyesha ni njia nzuri kiuchumi hasa ikiwa nyavu zitalenga kuvua samaki wakubwa nje ya maeneo yenye maji madogo ya ghuba.

Mazingira ya ghuba ya Chwaka yameonyesha kuwa na uwiano mzuri wa mfumo wa vyakula vinavyoliwa na samaki. Mfumo huo katika ghuba umejengeka kutoka ngazi ya chini ya aina gani ya vyakula vinavyoliwa na samaki kwenda juu, ukiwa na wingi wa samaki wanaotumia vyakula vya ngazi ya kwanza na ile ya pili na kuwa na wastani mdogo kwa ujumla. Madhara makubwa kwenye mazingira yameonekana kusababishwa na nyavu za kukokota pamoja na madema. Mitego hiyo kwa ujumla wake inaharibu mazingiza kwa njia wa kupunguza wingi wa aina za samaki wanaombatana na mazingira husika. Kwa ujumla mitego aina ya mishipi, nyavu za kukokota na madema ni mitego ambayo mavuvi yake hayana uwezo mkubwa wa kuchagua aina gani ya samaki wawapate wakati inapotumika kuvulia. Uvuvi wa kutumia madema umeonekana kuchangia sehemu kubwa ya mavuvi kiasi kwamba kati ya aina 6 za samaki wavuliwao aina 4 huvuliwa kutumia madema. Uvuvi wa nyavu za kukokota umeonekana kuwa na faida ndogo ijapokuwa unatoa fursa ya ajira kwa idadi kubwa ya wavuvi kwa kuwa ni aina ya uvuvi unaohitaji nguvu kazi kubwa. Kwa upande mwingine nyavu za kurambaza na majarife huweza kuvua kwa kuchagua aina ya samaki zimeweza kuwa na faida kwa mvuvi binafsi anayezitumia.

Kuigiza kwa kutumia njia mbali mbali imeonyesha ya kuwa uvuvi wa kukokota ukiondoshwa utasaidia kuongeza wingi wa samaki na faida. Hata hivyo mfumo huu unaweza kuwaacha wavuvi wasiopungua asilimia 58 % bila ya ajira na kwa hivyo kutokuwepo kwa njia mbadala na njia nyengine za kujipatia kipato na kuwa tegemezi kwa uvuvi na kwa hivyo sio suluhisho. Pia kuwachia tu watumiaji nyavu za kukokota si vizuri kwani utachangia kupungua kwa samaki zaidi hasa wale samaki wanaopendelewa kuvuliwa na kupoteza faida kwa wavuvi. Bila ya kuwa na muafaka wa faida za mvuvi mmoja moja (-20 %) na wingi wa viumbe hai katika mzingira (-30 %), nguvu za zana muhimu (i.e.madema, nyavu za kukokota na mishipi) hairuhusi maongezeko makubwa

(1.2 mpaka 1.4), lakini ongezeko linawezekana kwa nyavu za kurambaza, majarife na dhulumati (3.4 mpaka mara 4.2 . Lakini mtazamo huu unaweza kuwaacha zaidi ya wavuvi asilimia 37 % bila ya ajira

Kwa kumalizia inaonesha samaki wa Chwaka wamevuliwa kupindukia wakati aina ya samaki wengine wanaonesha kupungua. Changu (Lethrinidae) pamoja na samaki aina ya Chewa na Mzia wameweza kuvuliwa kwa njia endelevu kisiwani kote. Lakini kuwepo uvuvi uliokithiri kwa rasilimali za uvuvi Zanzibar haikuweza kuthibitika. Utafiti huu una changamoto mbili 1) wingi wa kuvuliwa samaki wadogo inamaanisha uvuvi usiokuwa endelevu na 2) matumizi ya zana haramu na kupunguwa kwa samaki wanaovuliwa ni viashiria tosha ya hali halisi ya rasilimali za uvuvi. Matokeo haya pamoja na kuwepo kwa taarifa kidogo za uhalisia wa rasilimali za uvuvi katika ukanda wa magharibi ya bahari ya hindi kunaonyesha kuhitajika kwa juhudi za kutathmini rasilimali za bahari ili kusaidia utoaji wa ushauri mzuri wa njia bora za udhibiti endelevu. Zaidi matokeo ya utafiti huu umeonyesha ya kuwa kuzuiliwa kwa matumizi ya uvuvi wa kukokota au kuleta aina nyengeni tofauti ya uvuvi ni jambo gumu kwa hivi sasa. Kutotambua uwezo wa boti zinazotumia nyavu za kukokota kuweza kuwa na wavuvi wengi na kuwatenga kikundi hiki utadhoofisha kutaarisha mpango mzuri wa kudhibiti matumizi yake. Ili kuondosha matuzi ya nyavu za kukokota Zanzibar usimamizi wa uvuvi ujikite katika kupunguza nguvu ya matumizi ya zana hii wakati huo huo kukiwa kunafanyika uwekezaji au uanzishaji wa njia mbada wa maisha.

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All these people really made this journey worthwhile.

CHAPTER I -
General Introduction

1.1. Small-scale fisheries with special emphasize on the Western Indian Ocean region

Small-scale fisheries are usually defined based on the size of the fishing unit, such as the size of the boats used or the number of operating fishermen. However, the term is often used synonymously with artisanal fisheries, which are mainly defined on the basis of the type of fishing gear used (i.e. relatively simple and traditional) and the low capital investment required (for details see FAO Glossary). Since artisanal fishing is generally small-scale with small boats and only a few fishermen operating, it is difficult to explicitly differentiate between these two fisheries. Thus, hereafter, we will refer to small-scale or artisanal fisheries as fisheries that 1) use small boats with low-technique propulsion (e.g. sail or low horsepower engines), which limits fishing activities to the inshore areas; 2) have low levels of capital investment; 3) are comprised of small fishing units; and 4) use a variety of high labour-intensive gears, which usually yield in a large diversity of target species (Batista et al., 2014; Salas et al., 2007).

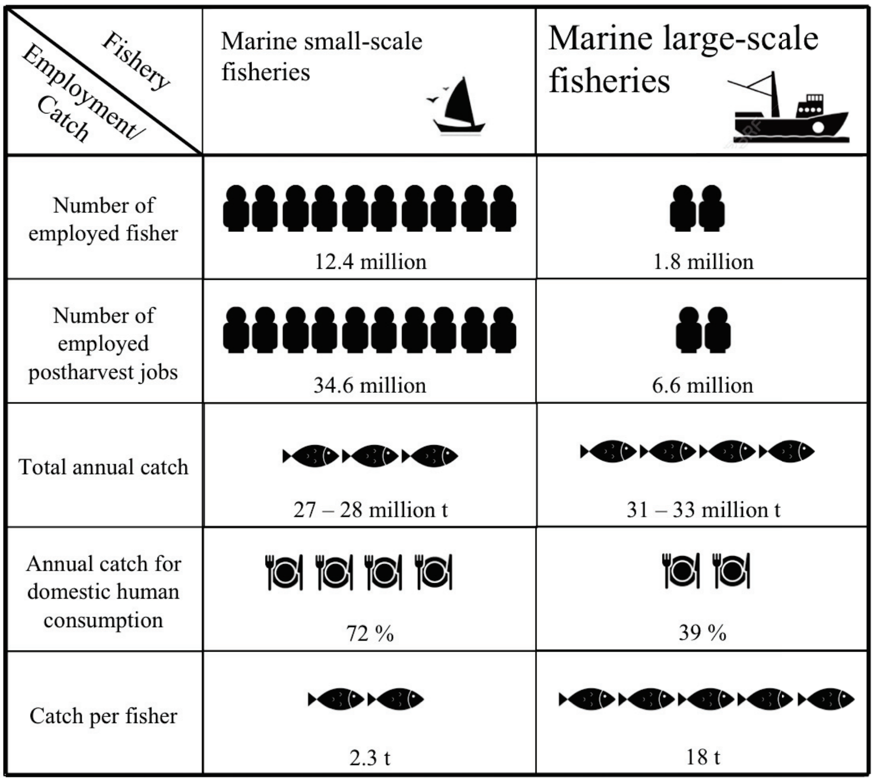

Fig. 1.1. Comparison of catch and employment of marine small-scale fisheries and large-scale fisheries. Figure was adopted from Pauly (2006) and updated with information provided in Mills et al. (2011).

Small-scale fisheries are major livelihood and animal protein provider around the world. A recent collaborative project of the World Bank, the FAO and WorldFish in 2010, estimated that 36 million full time and part time marine fisher operate in developing countries, which represents about 97 % of world's fishers (Mills et al., 2011). Likewise, estimates of total fish workers, including postharvest workers (e.g. fish processing), reveal that 97 % (118 million) operate in developing countries. The importance of marine small-scale fisheries for consumption and employment and its capture production compared to marine large-scale fisheries is depticted in Fig. 1.1. (Mills et al., 2011). In fact, small-scale fisheries are often a type of safety-net for low-income housholds, when failures occur in other livelihood strategies (Stobutzki et al., 2006). An estimated number of roughly 1.2 billion people live close to the coast line (100 km), and in many countries the coastal population accounts for the majority of the population (Nicholls and Small, 2003). It further has been estimated that the annual marine capture production of developing countries amounts to 61 million t of which 28 million t (46 %) is generated by small-scale fisheries (Mills et al., 2011, Fig. 1.1.). Only 39 % of marine large-scale catches is consumed by humans, while the large part of marine small-scale fisheries catches is used directly for local consumption (72 %, Mills et al., 2011, Fig. 1.1.). The low capital investment required in small-scale fishing sometimes provides the only opportunity for poor and vulnerable coastal inhabitants to get access to high animal protein (Kawarazuka and Bene, 2011).

This situation is very prevalent throughout the Western Indian Ocean region (WIO), which falls into one of the species richest marine regions on earth: the Indo-Pacific (Fig. 1.2., Wamsley et al., 2006; van der Elst, 2005). With its high variety of habitats and oceanographic conditions, the WIO region is home to a great faunal diversity and inhabits many endemic species (van der Elst et al., 2005). In total 2200 fish species are recorded belonging to 270 families (Smith Heemstra, 1986). This represents 15 % of global total marine fishes.

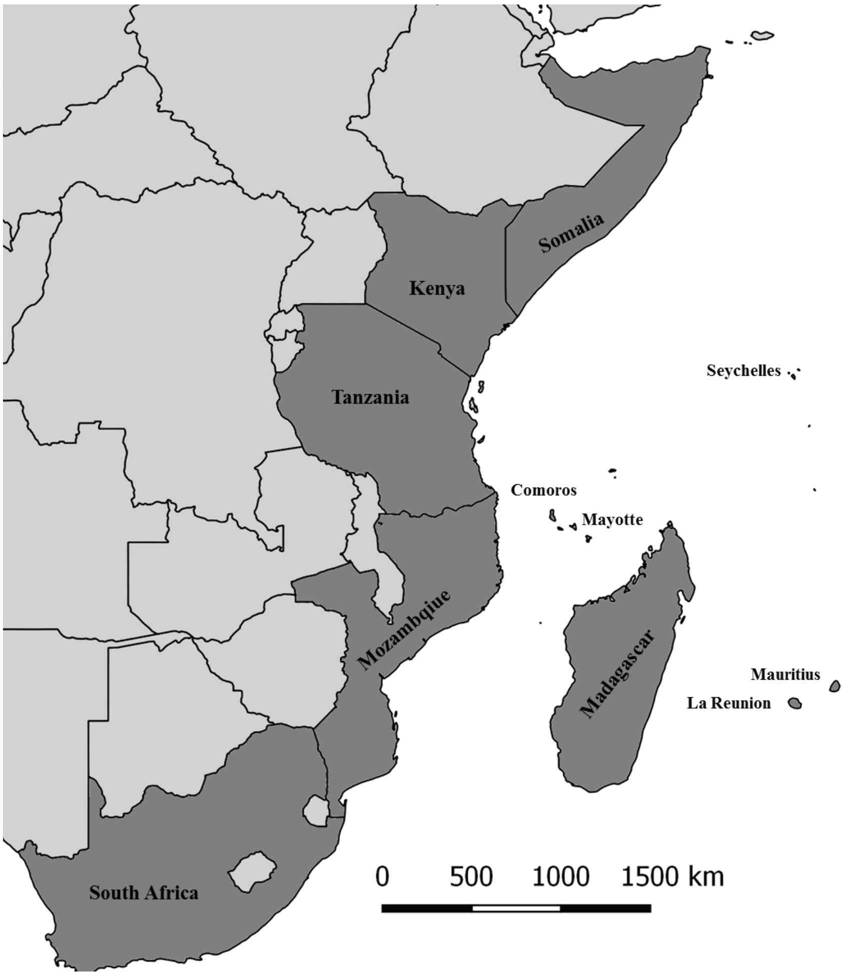

Fig. 1.2. Map of the Western Indian Ocean region

The fisheries in the WIO region are mainly of small-scale nature (Table 1.1.), except for South Africa[1], where only about 10 % of the fisheries is small-scale. Fisheries, including large-scale fisheries, can account for 5 to 99.1 % of food crops exports of the WIO countries (Walmsley et al., 2006). In Mozambique, for example, fisheries provide about 45 % of national exports (van der Elst et al., 2005). Overall, the fisheries sector contributes about 2 % to the overall GDP of the WIO region (Ardill and Sanders, 1991). Marine capture production is largest in Mozambique followed by Madagascar and Seychelles (Excluding South Africa[1], Table 1.1.). Likewise, in the WIO region fish is the prime protein source for a large part of the population (Walmsley et al., 2006). Not only is fish a more accessible protein source for coastal communities, it often possesses a much higher protein content than other animal protein sources of the

[1] Only part of the coast of South Africa belongs to the Western Indian Ocean, while the other part belongs to the South Atlantic Ocean. Capture production of South Africa is reported by the FAO for the entire coastline, hence South Africa was excluded from Table 1.1.

same monetary value (Walmsley et al., 2006). Small-scale fisheries are not only one of the major food providers in the region, but are often the prime livelihood of coastal communities (Barnes-Mauthe et al., 2013; Jiddawi and Ohman, 2002; Samoilys and Kanyange, 2008; van der Elst et al., 2005). In the Rufiji Delta (Tanzania), for instance, it has been estimated that 61 % of households are engaged in fishing (Walmsley et al., 2006). Most of the WIO countries are classified as developing and fall into the category of low income states. About 58.9 % of the population in the WIO region lives on less than 2 $ per day (Walmsley et al., 2006) and about one third of the population lives within 100 km of the coast (van der Elst et al., 2005), which further highlights the importance of fisheries for livelihood and food security in that region.

Table 1.1. Total capture production of 2015 in tonnes for all Western Indian Ocean countries (except South Africa) and the total marine capture production of 2015 obtained from the FAO official data. Information on the size of the SSF sector in each country was obtained from different sources (see below).

	Total capture production 2015 [t]	Marine capture production 2015 [t]	Contribution of SSF to the marine fisheries sector [%]
Unit. Rep.Tanzania	440433	96406	95[1]
Kenya	165321	8853	90[2]
Mozambique	286587	193567	83[3]
Madagascar	114754	88814	53[4]
Seychelles	102695	102695	NA
Somalia	30000	29800	60[5]
Comoros	12674	12674	95[6]
Mauritius	15505	15505	NA
Reunion and Mayotte	4812	4812	31[7]

[1]Jiddawi and Ohman, 2002; [2]USAID, 2016; [3]van der Elst et al., 2005; [4]Barnes-Mauthe et al., 2013; [5] http://www.fao.org/fi/oldsite/FCP/en/SOM/profile.htm; [6]FAO, 2015; [7]Le Manach et al., 2012

Despite the depth and distance limitations of small-scale fisheries, it has been demonstrated that they can have significant impacts on target populations and ecosystems (Hawkins and Roberts, 2004; McClanahan et al., 2008b, 1999). The overcapitalization of small-scale fisheries in particular has been identified as one of the biggest causes of overfishing of the respective fisheries resources (Pomeroy, 2012; Salayo et al., 2008; Stobutzki et al., 2006). This situation is largely caused by an open-access situation that is prevalent in many developing countries. In the WIO region it has been estimated that approximately 90 % of analysed fisheries has no regulation of effort (van der Elst et al., 2005). Poverty and the high dependency on fisheries resources each contribute significantly to the strong pressure on coastal systems. If resources are dwindling fishermen often respond by extending their fishing range, increase their time spent fishing, decrease their mesh sizes or switch to more efficient but often destructive gears (Kolding and van Zwieten, 2011; Teh and Sumaila, 2007). This in turn can lead to stronger pressure on coastal resources and likely also to more extensive declines in catch rates. This dilemma has been coined as Malthusian overfishing (Pauly, 1988) and has also been observed in the WIO region (e.g. McClanahan et al., 2008b). In fact, there is a strong concern of an unsustainable harvest of a number of different coastal resources (Jacquet et al., 2010; Le Manach et al., 2012; McClanahan et al., 2008b; Nordlund et al., 2013). Some of the target species that have experienced a particular strong decline in catches throughout the WIO region include some demersal fish, lobsters and sea cucumbers (van der Elst et al., 2005).

The lack of economic and political power of small-scale fishermen and the general focus of governmental support on industrial fisheries, leads to a strong marginalization (Allison and Ellis, 2001; Chuenpagdee et al., 2006; Salas et al., 2007). Furthermore, the general lack and/or low quality of information about target species and fishing effort (Batista et al., 2014; Salas et al., 2007; Salayo et al., 2008), complicates their assessment and management. For instance, when managing overcapacity, it is necessary to accurately estimate the number of active fishermen. However, it is difficult to monitor all fishermen, including full time, part time and occassional fisher. In addition, national surveys often disregard the participation of women and do not include them in the statistics (Kronen, 2002). In fact, the high complexity and heterogeneity of small-scale fisheries limits the ability of national fisheries institutions to accurately evaluate them (McClanahan et al., 2009a; Salas et al., 2007). This is worsened by a general low financial capacity and man power (Salas et al., 2007). This lack of data for small-scale fisheries is highlighted in the WIO region. Preliminary estimations of a World Conservation Union programme in East Africa (IUCN:EARO) revealed that only the shrimp fishery of Mozambique showed an adequate stock assessment and only 20 other fisheries had good levels of stock assessment, of which most were industrial fisheries from Mozambique (van der Elst et al., 2005).

The difficulty of monitoring and assessing small-scale fisheries together with the governmental focus on industrial fisheries, has created a situation where many small-scale fisheries are largely unregulated or where enforcement is poor (Stobutzki et al., 2006). In addition, the complexity of small-scale fisheries and the lack of institutional and financial capacities limit fisheries management in developing countries to the control of input measures (Pomeroy, 2012; Salas et al., 2007). Such measures include spatial and temporal closures, effort reduction as well as gear and mesh-size restriction. Particular, area and gear-based management forms the basis of most fisheries regulations within the WIO region (McClanahan et al., 2006, 2005a; Rosendo et al., 2011; Wells et al., 2010).

In the following sections I introduce the fishery of the study site in detail and discuss approaches how to assess tropical small-scale fisheries.

1.2. Artisanal fishery of the United Republic of Tanzania

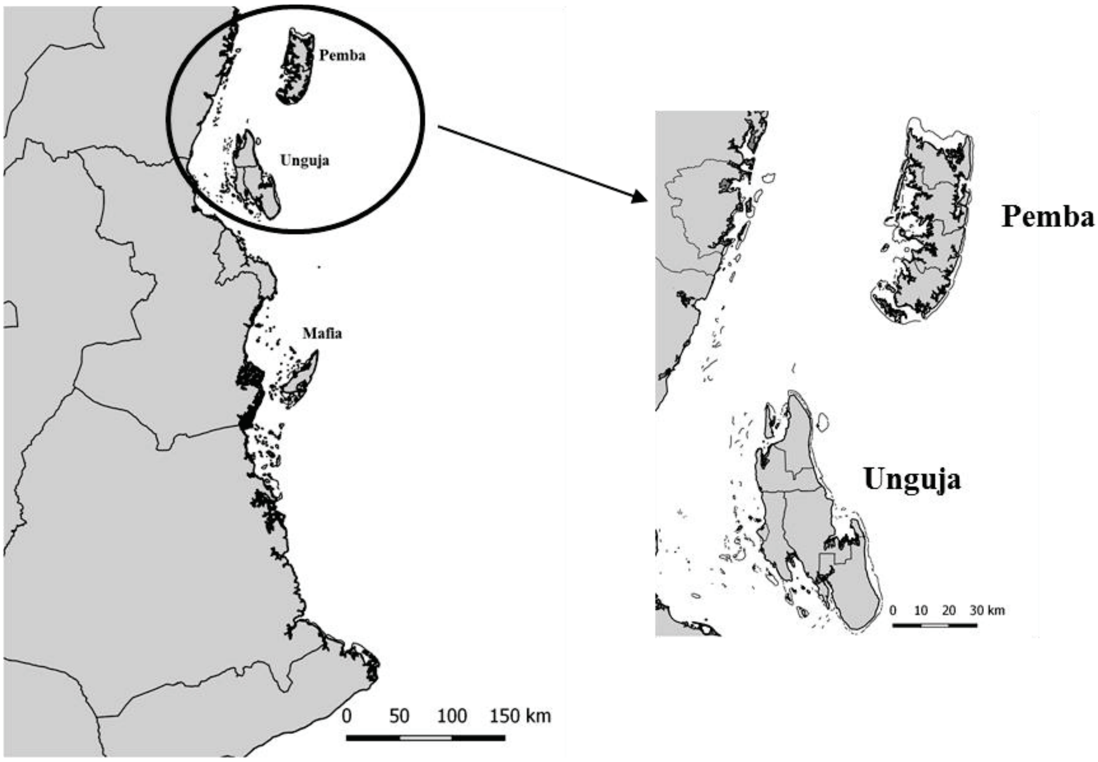

Fig. 1.3. Coastline of Tanzania mainland (left) and the islands of Zanzibar (right; Unguja and Pemba).

Tanzania is located on the east coast of Africa extending from latitude 4°49’S to latitude 10°28’S (Francis and Bryceson, 2001) consisting of the mainland and the near shore islands of the Mafia-Songo Songo archipelago and the islands of Zanzibar (Fig. 1.3.), The coastal area of the mainland covers about 30,000 km^2 (Gustavson et al., 2009). The semi-autonomous island state of Zanzibar consists of Unguja Island and Pemba Island, which are located about 40 - 60 km off the mainland. Both islands cover a total area of about 26,243 km^2 (Feidi, 2005). Around 25 % of the population lives in coastal areas (Gustavson et al., 2009). The population density and growth rate of Zanzibar is higher than on the mainland (January and Ngowi, 2010). Tanzania is known for a high marine biodiversity with 8000 species of invertebrates and 1000 species of fish (Francis and Bryceson, 2001). Most of the Tanzanian coastline, which is approximately 1424 km long (January and Ngowi, 2010), is surrounded by species rich coral reefs consisting of 150 different scleractian corals (Francis and Bryceson, 2001). Corals offer protection from wave energy for the coastline and provide habitat for a large number of marine species (Muthiga et al., 2008). Beside the coral reefs Tanzania possesses a great variety of different coastal environments including estuaries, mangrove forests, beaches and seagrass beds (Francis and Bryceson, 2001; Jiddawi and Ohman, 2002).

One of the most important activities for Tanzanian coastal communities is the inshore fishery and is one of the top three growth sectors (January and Ngowi, 2010) contributing 1.4 % to the GDP of the mainland (MLFD, 2013). On Zanzibar, however, the marine fishery plays a more prominent role having a share in GDP of 6 % (Lange and Jiddawi, 2009). The export value of marine fisheries resources for the mainland and Zanzibar is approximately 7,650,000 $ and about 598,000 $, respectively (Francis and Bryceson, 2001). Fish constitutes up to 30 % of the animal protein supply of the overall population of Tanzania and over 90 % of that of the coastal communities (January and Ngowi, 2010). Estimates suggest that Tanzania is home to approximately 55,000 fishermen and possesses approx. 400 landing sites (Jacquet et al., 2010). Only 20,000 fishermen are operating on the mainland, the greater part is active on Zanzibar (Jiddawi and Ohman, 2002; Feidi, 2005). The fishery sector on the mainland is mostly comprised of landings from lake fisheries as it possesses a great amount of freshwater resources through the lakes of Victoria, Tanganyika and Malawi. Approximately 80 % of the national fisheries production is attributed to the freshwater fishery (Fig. 1.4.). The annual marine catches in 2014 amounted to 97,072 tonnes, of which Zanzibar contributed with 33 % (Fig. 1.4., FAO).

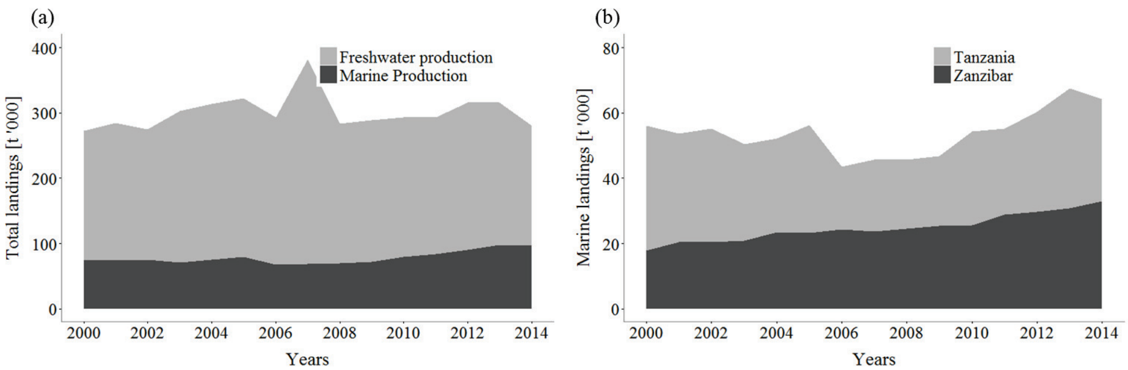

Fig. 1.4. Annual fisheries catches (t yr^{-1}) from 2003 to 2011. (a) Comparing marine and freshwater catches from Tanzania including Zanzibar and (b) comparing marine catches from the mainland with marine catches from Zanzibar. It can be seen, that freshwater fishery accounts for the greatest part of the catches (80 %) and that Zanzibar contributes significantly to the total annual marine catches (30 - 35 %). Data are provided by the FAO.

The fishing methods in use are mostly passive and are applied in depth not exceeding 30 m (Jiddawi and Ohman, 2002; January and Ngowi, 2010). Most common are hooks, lines such as handlines, longlines and troll lines as well as traps. Also common is the net fishery with a variety of different nets such as gill nets, trawl nets, cast nets, seine nets and dragnets[2]. Other gears used are spears, poison and dynamite (Jiddawi and Ohman, 2002). The catches are comprised of a variety of different marine species, including pelagic fish, reef fish, demersal fish and many invertebrate species (Jiddawi and Ohman, 2002).

The fishery on or adjacent to reefs accounts for approximately 70 % of the artisanal fish catch of Tanzania (Muhando, 2008). A highly important activity is the collection of invertebrates in intertidal areas around the whole coast, as it serves as an important food and protein source for the coastal households (Fröcklin et al., 2014). In addition, invertebrates are the prime marine export products of Zanzibar, including lobsters prawns, sea cucumbers, seashells, crabs, octopus and squids (January and Ngowi, 2010; Jiddawi and Ohman, 2002).

The restriction of the fishery to near shore areas is partly due to the lack of money and technical skills, which leads to poor cooling systems and a small percentage of motorized boats (January and Ngowi, 2010). Often the fish cannot be stored in freezers due to a lack of electricity, thus the quality of the fish decreases and therewith its market value (Jiddawi, 2012). In 2001 there were a total number of 4927 vessels operating in the waters of Tanzania (Sobo, 2004), of which only 513 boats were motorized (Jiddawi and Ohman, 2002). Often fishermen do not possess the financial means to buy and maintain the required fishing equipment and hence enter a partnership with so-called middlemen/patrons, who own the vessels and/or gears (Ferrol-Schulte et al., 2014). The income is then split into three parts, one for the middlemen, one for the maintenance of the boat and one for the fishermen (de la Torre-Castro and Rönnbäck, 2004; Jiddawi and Ohman, 2002). Another important aspect of the Tanzanian fishery is the presence of migrating fishermen, which results in a temporal increase of fishing effort in certain areas and can cause problems in the management of fisheries (Jiddawi and Ohman, 2002; Wanyonyi et al., 2016; Mildenberger, 2015).

[2] Similiar to beach seines, but are applied in the intertidal area using boats.

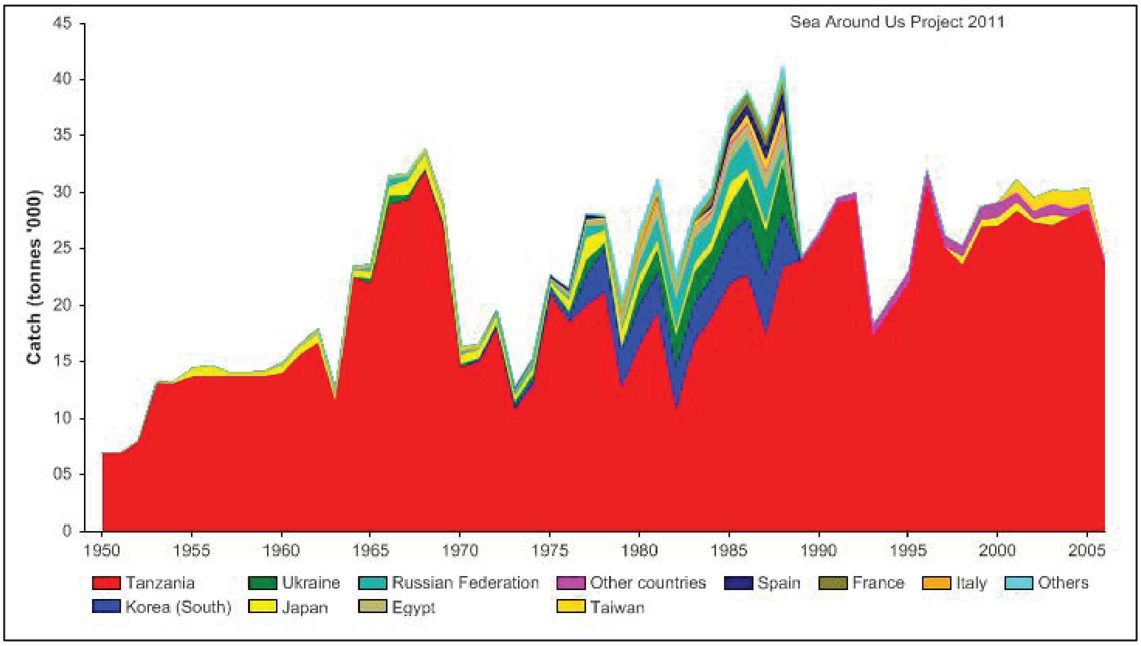

Fig. 1.5. Annual catch (tonnes '000) for the Tanzanian Exclusive Economic Zone in 2011. Figure was obtained from the "Sea around Us Project" (www.seaaroundus.org).

Due to the artisanal character of the Tanzanian fishery the exclusive economic zone (EEZ), an area of 241,541 km^2, is little exploited with an annual catch of approximately 30,000 tonnes in 2005 (Feidi, 2005; Jiddawi and Ohman, 2002; www.seaaroundus.org). This suggests a potential for further development of the fishery sector with subsequent increased employment. However without detailed knowledge about standing stocks it is impossible to determine the exploitation status of these stocks and its potential for marine fisheries of Tanzania. The industrial fishery in the waters of the EEZ started in 1960 with vessels mostly financed or operated by European countries (Jacquet et al., 2010). In 1998 Tanzania started to grant licenses for fishing activities in these waters and the number of vessels increased to 64 in 2004 (January and Ngowi, 2010). By the end of the 80s the total catch by foreign countries decreased to zero and remained low (Fig. 1.5., www.seaaroundus.org). The highest total catch of the EEZ was recorded in 1988 with approximately 41,200 tonnes and decreased in 2006 to a value of approximately 23,600.

As the fishery of Tanzania is open-access (Sobo, 2004) and hence anyone can participate, the amount of fishermen has increased over time (Jiddawi and Ohman, 2002). Despite this increase in fishing effort, the Tanzanian inshore catches are said to show a decreasing trend, which indicates an overexploitation of the fisheries resources (Jacquet et al., 2010; Jiddawi and Ohman, 2002; Mkenda and Folmer, 2001; Phelan and Stewart, 2008), causing the need for a development and improvement of the fisheries resource management (cf. annual catch statistics in Fig. 1.6., Jiddawi and Ohman, 2002). In addition, the degradation of Tanzanian coral reefs is attributed to inter alia overfishing and the use of destructive fishing gears (Muhando et al., 2002; Muthiga et al., 2008). One of these fishing methods is the dragnet fishery, which applies seine nets from either the beach or in coral reefs by dragging them along the bottom and thereby damaging bottom dwellers, seagrass beds and corals (January and Ngowi, 2010; Mangi and Roberts, 2006). Other destructive fishing methods encompass prawn trawling, dynamite, spear guns and poison (Jiddawi and Ohman, 2002). The dynamite fishery was banned in the 1990s, but returned and is currently experiencing an increasing trend (Muthiga et al., 2008; Slade and Kalangahe, 2015). Another prevalent issue is the use of very small mesh sized nets, which catch not only small species but also the juvenile part of target populations (January and Ngowi, 2010). This particularly applies to mangrove and seagrass areas as they represent nursery grounds for many fish (Ngusaru et al., 2001).

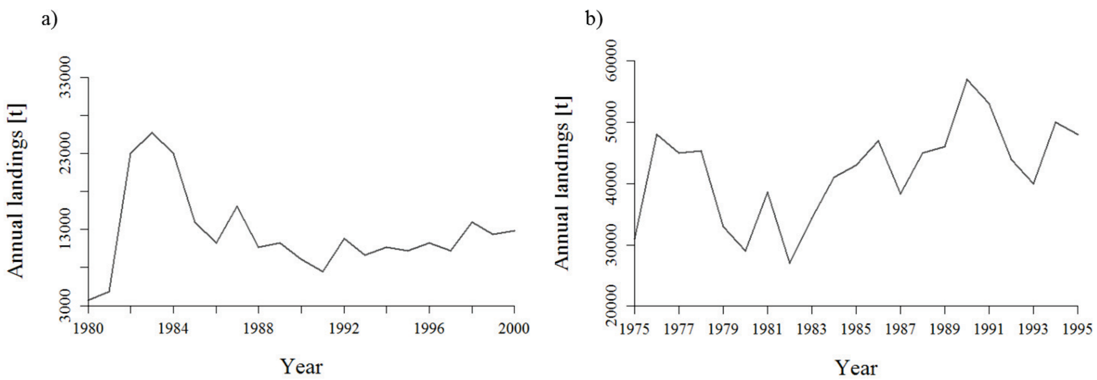

Fig. 1.6. Total annual landings of a) Zanzibar from 1980 to 2000 and b) Tanzania Mainland from 1975 to 1995. Data was obtained from Jiddawi and Ohman 2002.

Despite Tanzania's activity in a number of regional and international fisheries management bodies, Tanzania still fails in the attempt to reach its aims and goals set under the framework of a sustainable resource management (January and Ngowi, 2010). Currently most of the knowledge on fisheries stocks and the biology of target species is collected through so called beach recorders, which are officially employed by the Department of Marine Fisheries Resources (DMFR) and sent to the landing sites for 16 random days per month (Jiddawi and Ohman, 2002). The collected fisheries data encompass mostly the total annual catch in kg and price per target family. Tanzania also possesses a coral monitoring program, which was implemented in the late 1980s. In addition scientific studies and surveys are carried out irregularly in order to assess the status of different fisheries (e.g. Mkenda and Folmer, 2001; Jiddawi and Ohman, 2002; Lokina, 2002; Eriksson et al., 2010). Despite these programs and surveys, the lack of information on the biomass of target species remains a major problem. Knowledge on stock size and age composition, for instance, is missing for the majority of the target species and the spatial resolution of existing fishery data is still very low (Jiddawi, 2012; January and Ngowi, 2010). Furthermore mortality and growth rates are still missing (Sobo 2004) for the calculation of sustainable yields and hence for the development of sustainable harvesting strategies; likewise independent ecological data for most of the marine species are absent (Jiddawi and Ohman 2002).

Another aspect leading to poor management of Tanzanian fishery resources is the poor implementation and enforcement of regulations. The Tanzanian Fisheries Act of 1988 defined a legal size for 23 fish species (Phelan and Stewart, 2010) and the Zanzibar Constitution 1984 Order prohibited the collection of undersized (< 100 mm) sea cucumbers (Eriksson et al., 2010). Furthermore, in the Zanzibar Fisheries Act a minimum mesh sizes is defined (1.5 inch) and the use of destructive fishing methods (i.e. dynamite, poison, spear guns, beach seines) is prohibited (RGZ, 2010). However these restrictions and regulations are not well enforced and often fishermen do not obey these prohibitions (Jiddawi, 2012). In addition to these fisheries regulations the government of Zanzibar tries to encounter the problem of unsustainable fisheries resource use by the implementation of marine protected areas and the promotion of more sustainable marine-based activities such as mariculture, value-adding activities such as ecotourism as well as alternative livelihoods (Lange and Jiddawi, 2009). The promotion of activities in the tourism sector is intended to release some of the pressure on the fisheries resources, since “The economic importance of tourism is five times the combined size of other ecosystem values.” and hence it is a sector, which would provide stable occupation and a safe income for the coastal people of Tanzania (Lange and Jiddawi, 2009).

An essential aspect in the fishery of marine resources is the strong role of coastal communities in the implementation of management plans. Jiddawi (2012) points out that communities need to be involved in the process and establishment of management plans in order to ensure the success of a sustainable use. This is obvious as coastal communities rely heavily on fisheries resources and consequently might disobey regulations if they do not recognize their benefits or simply to ensure subsistence. The Department of Marine Fisheries Resources therefore initiated the formation of community-fishermen committees in all fishing villages, the so-called Village Fishermen Committees (VFC) in order to implement management strategies (Jiddawi, 2012). However, due to a more centralized management system, the village committee lacks essential power and capacities in their scope of action. The beach recorders often have strong relationships with the local fishermen, which further limits their ability to monitor fishing activities and enforce regulations (de la Torre-Castro, 2006). The main reason for the missing knowledge and the failure in the implementation of regulations is the lack of human, technical as well as financial resources (Sobo, 2004; January and Ngowi, 2010). In addition the fisheries sector is not governed in a holistic and integrated way (January and Ngowi, 2010), which hinders its effective management.

An added complication is that the Tanzania mainland and Zanzibar each have autonomous institutional and legal structures for managing fisheries (Jacquet et al., 2010). In conclusion, the missing knowledge of the status of the fishery, the poor implementation of management plans and the absence of proper surveillance together with several obstacles such as the lack of financial means, high dependency of the coastal communities on fishery resources and kinship structures impedes the successful management of Tanzanian fisheries resources substantially.

1.3. Chwaka Bay

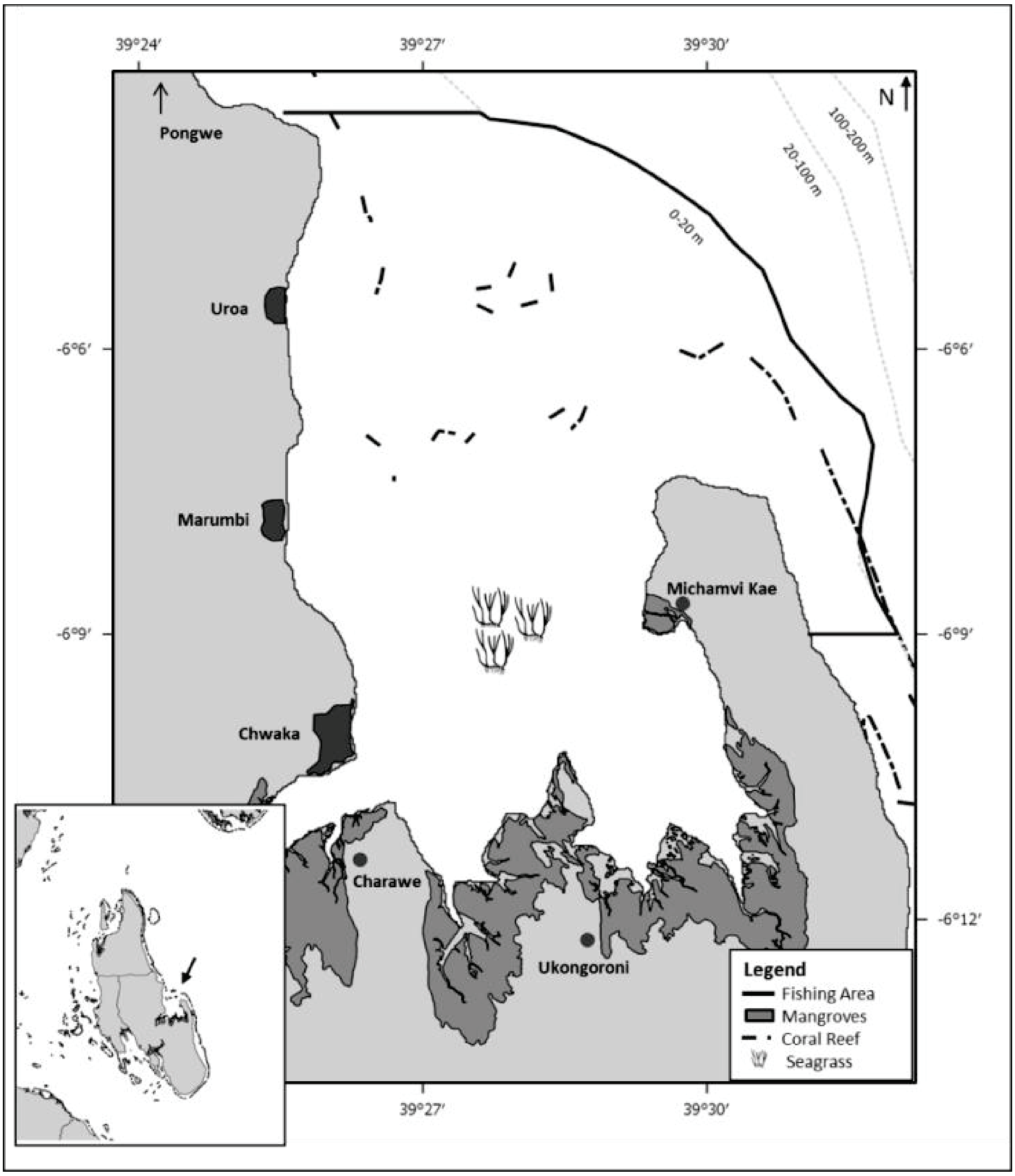

Fig. 1.7. Chwaka Bay, Zanzibar (Tanzania). The bay is comprised of large mangrove stands in the south and a fringing reef at the bay opening. Seagrass meadows are found throughout the entire study site with dense aggregations in the bay proper. The extension of the study area is depicted by a solid line.

Chwaka Bay is located on the east coast of Zanzibar and consists of mangrove, seagrass and coral reef habitats (Fig. 1.7.). The bay proper covers an area of 50 km^2 with an average depth of 3 m and a maximum depth during high tide of 5 m. During lowest spring tide approximately 60 % of the bay is exposed and during neap tide approximately 90 % of the water is renewed (Shaghude et al., 2012). The reef protects the bay from wave energy and deters the mixing of bay water and oceanic water. Surface water temperatures range from 25 - 31° C (Jiddawi and Lindström, 2012). Despite the lack of freshwater bodies surrounding the bay, the average salinity is 26 ‰ in the inner and 35 ‰ in the outer parts of the bay (Jiddawi and Lindström, 2012). This is probably explained by the groundwater seepage and minor seasonal streams (Shaghude et al., 2012).

Exceeding the importance of seaweed farming in regard to income (Eklöf et al., 2012), fishing is the most important source of income for the population of Chwaka Bay (Jiddawi and Lindström, 2012). For more than 50 % of the population it is the only source of income, while the other 50 % is involved in fisheries indirectly to make a living (Jiddawi, 2012).

Fish products contribute up to 98 % to the animal protein supply for the people of Chwaka Bay and the per capita fish consumption is approximately 17 kg, more than the world average fish consumption of 16.2 kg (Jiddawi, 2012). Seven fishing villages are found in Chawka Bay: Pongwe, Uroa, Marumbi, Chwaka, Michamvi, Ukongoroni and Charawe, with Chwaka and Uroa being the biggest and most important landing sites (Jiddawi, 2012). Fishermen´s earnings range between 1 $ to 20 $ per day. About 10-20 % of the fish catch is used for own consumption, while the rest is sold for export or to hotels and restaurants (Jiddawi and Lange, 2009).

In Chwaka Bay the most common fishery is the basket trap fishery called "dema" (52 %), the handline fishery (26.6 %) and the net fishery (Jiddawi, 2012; de la Torre-Castro and Lindström, 2010, for more details about relative gear composition at the different landing sites see Fig. S.1.1.). However the net fishery has only recently been introduced, as it is quite expensive and (particularly the dragging procedure) requires a motorized vessel and fuel (de la Torre-Castro and Lindström, 2010). Almost 50 % of the vessels used are still outrigger canoes (Jiddawi, 2012).

The catch of the Chwaka Bay fishery consists of more than 200 different fish species (Jiddawi, 2012). An overview about the catch composition of the Chwaka Bay fishery can be found in Fig. S.1.2., which is based on sampled catches from all different gears obtained in 2014. However not only fish is an important marine resource for Chwaka Bay, also the crab fishery contributes significantly to the livelihoods and protein supply of the population. Other marine resources harvested include sea cucumbers, gastropods, bivalves and lobsters. According to the qualitative judgement of Hamad Kathib (pers. comm., May 2013) around 90 % of the annual landings originate from the mangrove and seagrass areas, thus the reef fishery in Chwaka Bay is of no great significance.

The number of fishermen in Chwaka Bay has increased from approximately 1469 in 2003 to approximately 1871 in 2007 (Fig. 1.8.), thereby demonstrating a clear trend towards an increase in fishery engagement (Jiddawi, 2012). Despite the increase in fishing effort, the catches significantly declined over time. In 1990 the total catch was approximately 950 t and has declined to 370 t in 2004 and then progressively to about 320 t in 2007 (Fig. 1.8.; Jiddawi, 2012). This overall trend indicates an overexploitation of the resource (Jiddawi and Ohman, 2002; Jiddawi, 2012). One major problem of Chwaka Bay is that the area fished, mainly consists of nursery and breeding grounds of many fish species and fisher use very small mesh sized-nets which, together result in the capture of undersized, juvenile fish (Lugendo, 2007; Jiddawi, 2012). Another problem is the ongoing use of destructive fishing methods in the bay, particularly the use of dragnets.

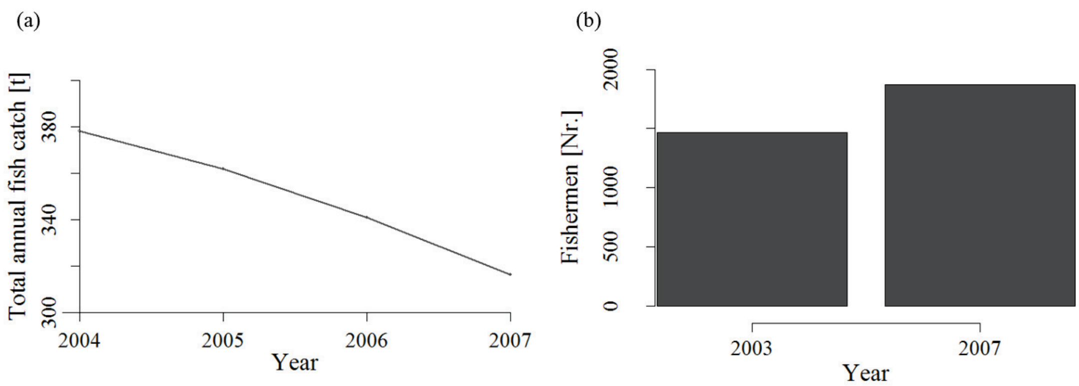

Fig. 1.8. a) Fish catch trend from 2004 to 2007 at Chwaka Bay and b) number of fishermen fishing in Chwaka Bay for 2003 and 2007 (source: Jiddawi, 2012).

1.3.1. The Chwaka Bay seagrass ecosystem

Seagrass beds are a typical feature in Tanzania, distributed widely from intertidal to subtidal areas (Francis and Bryceson, 2001). Chwaka Bay has an exceptional diversity of seagrass species with 11 different species, of which *Enhaulus acroides*, *Thalassia hempriichi*, *Cymodocea rotundata*, *Cymodocea serrulata* and *Thalassodendron ciliatum* are the most dominant ones. Seagrass beds can be found in large assemblages or in fragmented small patches mostly occurring in communities of different seagrass species and macroalgae species, such as *Halimeda spp.* (Gullström et al., 2012). Seagrass beds are one of the most productive marine ecosystems in the world (Duarte and Chiscano, 1999) and provide a large range of goods and services (Gullström et al., 2002). They are essential for the stabilization of sediments and act as a trap for nutrients and organic matter in the bottom sediment (Fonesca, 1989). Further the primary and secondary production in these heterogeneous habitats is very high (Gullström et al., 2002). Most importantly seagrass beds serve as nursery, breeding and feeding habitats for many marine species (Bell and Pollard, 1989). In Chwaka Bay it has been shown by Dorenbosch (2006) that seagrass meadows show a high interlinkage with the adjacent coral reef; further Gullström et al. (2012a) demonstrated that their significance for reef dwellers exceed that of mangrove forests. In general seagrass beds show higher abundances, biomass and diversity than unvegetated areas (Eklöf et al., 2005) supporting a large variety of different taxa ranging from microorganisms over benthic fauna to fish (Gullström et al., 2002). In Chwaka Bay 35 different families of fish could be identified by Gullström et al. (2008), with Apogonidae, Blenniidae, Centriscidae, Gerreidae, Gobiidae, Labridae, Lethrinidae Lutjanidae, Monacanthidae, Scaridae, Scorpaenidae, Siganidae, Syngnathidae, and Teraponidae being the most common fish taxa (Gullström et al., 2002) and *Leptoscarus vaigiensis* being the most abundant species (Gullström et al., 2008). Due to the high primary productivity and the rich species assemblage, seagrass beds are one of the most important fishery areas (de la Torre-Castro et al., 2014). As seagrass beds are mainly found in shallow waters they suffer of large human activities resulting in a clear decline of dwelling species and the destruction of the seagrass itself (Eklöf et al., 2006; Gullström et al., 2002). The major causes of seagrass loss are the eutrophication of coastal waters, destructive fishing practices (dragnet fishery), sediment alteration, coastal construction and seaweed farming (de la Torre-Castro and Rönnbäck, 2004; Delgado et al., 1999). Especially the use of dragnets is highly destructive, as the net is dragged over the substrate, causing severe damage (Gullström et al., 2012).

1.3.2. The Chwaka Bay mangrove ecosystem

The mangrove forests of Tanzania cover an area about 123,500 ha and 18,000 ha of it is located on Zanzibar. In total there are 10 different mangrove species distributed over the mainland and Zanzibar (Lugomela, 2012). Chwaka Bay contains the largest mangrove area on Unguja Island (Lugomela, 2012); the forest covers 90 % and 50 % of the southern and eastern coastal stretches of the bay and inhabit all ten species (Jiddawi and Lindström, 2012). Mangroves are highly productive ecosystems and offer protection and stabilisation of shorelines (Lugomela, 2012). Their high turnover rates and the accumulation of suspended organic matter make them essential for water quality, particularly in adjacent habitats such as seagrass meadows and coral reefs (Ngusaru et al., 2001). Mangroves contribute to the total riverine organic carbon input in the open ocean with an estimate of 10 %, however in Chwaka Bay most of the exported mangrove material is trapped in the seagrass beds and hence does not leave the bay (Lugomela, 2012). Mangroves provide feeding and nursery grounds for many marine species of fish, bivalves, gastropods, prawns and crabs (Ngusaru et al., 2001), while for other species such as oysters and crabs it serves as a permanent living environment (Franics and Bryceson, 2001). Lugendo et al. (2007) identified 150 fish species out of 55 families in the mangroves of Chwaka Bay. It is therefore a major economic resource and contributes substantially to the fishery catches (Jiddawi and Lindström, 2012). One of the major fisheries practiced in mangrove areas is the prawn and crab fishery, which is also extensively conducted in the area of Chwaka Bay, particular Ukongoroni (Pereira et al., 2009). Chwaka Bay mangroves inhabit a variety of different crabs of which most are edible and are therefore increasingly targeted (Pereira et al., 2009). A major species is the mud crab *Scylla serrata*, which is collected by hand during low tide. Jiddawi (2012) states that about 44 % of the fishers catch 4 - 10 crabs per day. Other mangrove species harvested are shellfish such as oysters and cockles. The common fishing gears used in mangrove areas are traps and gillnets (Pereira et al., 2009). Despite the high fishing pressure, mangroves are also subject to the cutting of mangrove trees, which is a vast business in Chwaka Bay, since the wood is used for a variety of different purposes including firewood, wood for construction, vessels and fuel (Ngusaru et al., 2001; Jiddawi and Lindström, 2012). These activities in combination with the conversion of mangrove areas into salt pans and the clearance for urban and industrial development (Francis and Bryceson, 2001) result in a strong destruction of mangrove forests with a loss rate exceeding that of rainforests and coral reefs (Duke et al., 2007).

1.4. Assessments of tropical small-scale fisheries with emphasis on the use of an ecosystem-based approach to fisheries

The fundamental aim of fisheries management is to ensure the indefinite exploitation of target resources at a close to optimum production level. Because fishing effort cannot be increased unlimited without eventually compromising a given stock and hence decreasing its respective production potential and yield, fisheries managers need to know the optimum fishing pattern and level to generate a desired yield (mostly, the long term maximum sustainable yield, MSY, is envisioned). In this regard, fisheries stock assessment aims at informing managers about the exploitation status of target resources and providing advice on thresholds and ranges of optimum exploitation. Thereby stock assessment functions as the process that links the input of a given fishery (i.e. fishing effort) to the output (i.e. catch of fish landed) of that fishery. However, one of the central problems in the management of tropical small-scale fisheries is the lack of data, which limits the number of methods that can be used in the evaluation of target stocks (Johannes, 1998; Honey et al., 2010; Batista et al., 2014; Sparre and Venema, 1998). In conventional stock assessment of tropical fisheries two main groups of methods are used, holistic (“biomass pool”) and analytical (“size/age structured”) models (Sparre and Venema, 1998). Holistic models are relatively simple and are based solely on time series of catch and effort data. They are holistic in that they consider a fish population as a biomass pool, rather than accounting for differences in biomass and life history characteristics across different size/age classes (cohorts). Such methods include surplus production models, which have widely been applied in data poor fisheries, since they only need catch and effort data to evaluate fisheries (Maunder et al., 2006). Nevertheless, such methods still require reliable information on fishing effort, a factor that is hard to estimate in small-scale fisheries, given the flexible behaviour of fishermen, their spatially scattered fishing grounds and the lack of institutional capacity.

A very simple but crude method is the catch-based method (Froese and Kesner-Reyes, 2002). This method analysis trends in catches of a target population over time in relation to its historical maximum catch. However, it should be applied to a) relatively long time-series data and b) data recorded at species level. The reported catches in the WIO region are aggregated to family or group level, which minimizes the utility of this method.

The analytical models (Sparre and Venema, 1998) are more data demanding than holistic models but have the advantage that they take into account the age/size structure of a stock. Accordingly, they require the knowledge of the size/age frequency distribution of a given population usually obtained from catch data. These models are strongly based on "Virtual Population Analysis (VPA)" and "yield/recruit prediction models". While VPA serves to estimate the fishing mortality and biomass of each length/age group of a given stock, thus providing a comprehensive picture of the current stock size and fishery status, prediction models are used to explore the effect of different fishing patterns on target stocks in the future. These models require an understanding of the stock-specific growth and mortality characteristics. Such information can also be used by managers to estimate the vulnerability of target stocks relative to each other and implement management measures accordingly (Honey et al., 2010). The estimation of total mortality (e.g. using linearized catch curves) together with estimations of natural mortality provides managers with information on the current level of exploitation of their target species. The growth characteristics required are usually obtained through age-readings of otoliths or fish scales or through the analysis of length-frequency samples. The advantage of the presented analytical models in the tropical context is that they do not require an accurate estimation of effort or catch, which are difficult to obtain. Measuring the length-frequency distribution of key species from a well-structured subsample of catches is relatively cheap, easy and generates a data set with lower uncertainties.

However, these methods are based on the use of single stock dynamics. Particularly in the tropical context, such models are insufficient to represent the dynamics of local fisheries, as they use a variety of different fishing methods catching a great diversity of species. While a mesh-size control might be useful to protect the target populations of single-species fisheries, in multispecies fisheries it will most probably lead to the under- or overexploitation of the majority of target stocks (Pauly, 1979). Furthermore, such methods can only say something about the status of target species and do not include non-target species let alone whole ecosystems. It has been shown that fisheries can have significant influences on the biodiversity (Myers and Worm, 2003) and the trophic structure (Pauly et al., 1998) of ecosystems. Fishing impacts on target species can propagate through the food web and induce indirect changes on non-target species, such as the proliferation of prey species that might eventually lead to regime shifts (Österblom et al., 2007). Such indirect effects are induced through species interaction, of which natural predation has shown to be one of the key interactions in fish communities (Hixon, 1991). Thus, a fishery could be considered sustainable in a single-species context, while from an ecosystem perspective it may already be overfished. Furthermore, some fishing techniques might also have a strong impact on habitat architects such as corals and seagrass beds, which in turn can have devastating impacts on target species (Beukers and Jones, 1998). Thus, while single species methods are useful to understand the status of the main key species or identify the most impacted and vulnerable stocks, holistic management of sustainable fisheries resources calls for an ecosystem-based approach to fisheries management (EBFM). Pikitch et al. (2004) identified four key components that should constitute an EBFM approach:

i. *avoid degradation of ecosystems, as measured by indicators of environmental quality and system status;*
ii. *minimize the risk of irreversible change to natural assemblages of species and ecosystem processes;*
iii. *obtain and maintain long-term socioeconomic benefits without compromising the ecosystem;*
iv. *generate knowledge of ecosystem processes sufficient to understand the likely consequences of human actions.*

In fact, the use of the ecosystem-based approaches becomes increasingly imperative in the framework for the management of fisheries (Hall and Mainprize, 2004). The FAO, for instance, produced guidelines on the ecosystem approach to fisheries in order to supplement the FAO Code of Conduct for Responsible Fisheries (Garcia et al., 2003).

However, it is an enormous task to evaluate all possible fishing impacts on a given ecosystem. Several quantitative approaches have emerged to help understanding the dynamics and interactions between different ecosystem components and ultimately detecting their nonlinear responses to artificial stressors such as fishing (Collie et al., 2014). As elaborated above, one of the major dynamics within fisheries resource communities is the predator-prey relationship. Hence quantitative approaches to ecosystem-based fisheries management need to integrate trophic interactions (Collie et al. 2014). Furthermore, in the context of an EBFM the central focus must lie on fisheries management and outcome must be able to inform managers (Collie et al., 2014). Good examples of such approaches are the ecosystem models *OSMOSE* (Shin and Cury, 2001), *Ecopath with Ecosim and Ecospace* (*EwE*, Walters et al., 1997) and *Atlantis* (Fulton et al., 2011). However, a strong weakness of most ecosystem models is that the model itself is not fitted statistically to the input data; nor do they give confidence intervals based on model uncertainty (Hill et al., 2007). In *EwE* this is partly addressed through fitting processes of key parameters to time-series in *Ecosim* and by the *Ecoranger* routine, which finds alternative parameter combinations for the model (Walter et al., 1997).

In fact, with the development of the free *EwE* software, a useful tool has been created that allows for the holistic and integrated analysis of aquatic systems and the simulation of spatio-temporal responses to different fishing strategies (Polovina, 1984; Walters et al., 1997; Christensen et al., 2008). It offers the user a means to integrate different functional groups of a system ranging from primary producers to top predators, their feeding interactions as well as their spatial constraints. Furthermore, the user can explore the interaction between bottom-up and top-down (particular fishing) effects on the system. Additional plug-ins such as the fishing policy search have been developed for the exploration of fishing scenarios and ultimately to aid in the management of fisheries (Christensen et al., 2008). *EwE* is still of relative simplicity and not too data demanding (Heymans et al., 2011). This is particularly important in a tropical fisheries context, due to the poor data availability. More importantly, the extensive and long-term use of the software has generated a compilation of models ranging from the tropics to temperate regions, including all kinds of different ecosystems such as seagrass meadows, coral reefs and deep seas (Heymans et al., 2014). Accumulated knowledge from all these models can be used in model construction and comparisons of model outcomes. What makes this *EwE* ecosystem modelling approach particularly suitable for ecosystem-based fisheries management in small-scale tropical fisheries is that it allows for accounting of fisheries related socio-economic factors (e.g. provision of jobs and profits, Christensen et al., 2008). Due to the high dependency of tropical fishing communities on fisheries resources, it is imperative that approaches to EBFM include the effects of different fishing strategies or conservation plans on the local community. This is not only crucial for the compliance with regulations but for food-security and poverty alleviation. Overall, *EwE* is a unique, highly useful tool for fisheries management to move towards an ecosystem-based approach.

1.5. Scope of the thesis and thesis outline

1.5.1. Thesis objectives

The present dissertation stems from the need for a proper assessment of Zanzibar's and in particular Chwaka Bay's fisheries resources in order to provide fishermen and managers with comprehensive information for appropriate management plans. The reason to choose Chwaka Bay as a reference site was largely based on its high ecological diversity, the market at Chwaka village, which is one of the most important markets on the Island and the strong concern for an unsustainable resource exploitation. Furthermore, the bay represents an ideal study area for ecosystem modelling due to its ecological and topographic (e.g. semi-enclosed system) features together with the presence of a relatively high amount of quantitative information about the benthos and primary producers of the bay. In addition, this study aims to provide insights into the small-scale fisheries of the WIO region and to contribute towards better solutions for their management, since many WIO countries show similar fisheries settings.

The scope of this thesis is, therefore, to assess the fishery of Chwaka Bay by applying single-species methods and by placing the fishery into an ecosystem-based context using the *EwE* ecosystem model. Furthermore, using Chwaka Bay as a reference site the thesis aims at approaching the answer to the question of the sustainability of Zanzibar's nearshore fisheries. The following research questions (RQ) were addressed in the different chapters:

I. Where is the evidence for an overexploitation of Zanzibar's nearshore fisheries and to what extend are they overexploited?
II. What is the state of selected key target species of the Chwaka Bay fishery?
III. What is the overall state of the Chwaka Bay ecosystem and its fishery? How are the different gears in use impacting the trophic structure and energy flows of the system? What is the impact of the different gears on the local fishing community?
IV. What are the impacts of a reduction or reallocation of dragnet fishing effort on the biomasses of the different functional groups and on the profits of the fishing communities? How can the dragnet fishing effort be reduced without compromising the livelihood of the fishing community?

1.5.2. Thesis outline

This dissertation is comprised of six chapters. In Chapter I, I put this study into context by reviewing the literature on the importance of small-scale fisheries with emphasis on the WIO region and I provide an overview of the fishery in Tanzania and the study site Chwaka Bay. Furthermore, I address the complexity of tropical small-scale fisheries management and discuss approaches for their assessment.

Chapter II comprises a literature review which identifies and quantifies the evidence of overexploitation on Zanzibar in order to answer RQ I. Furthermore, this chapter contains an evaluation of Zanzibar's overall landings between 1990 and 2014 using the catch-based method (Froese and Kesner-Reyes, 2002).

In Chapter III, the catch length frequency composition of six of the main target species (i.e. *Siganus sutor, Lethrinus borbonicus, Lethrinus lentjan*, *Lutjanus fulviflamma*, *Leptoscarus vaigiensis* and *Scarus ghobban*) is assessed over an annual cycle in order to answer RQ II . Firstly, the length-frequency distributions of the six key species are used to estimate juvenile retention rates and derive growth characteristics with the help of *ELEFAN I* as implemented in the program package *FiSAT II* (Gayanilo et al., 1994). Second, stock size as well as current fishing mortalities and exploitation rates across size-classes are estimated using the length-based catch curve and length-based Jones cohort analysis (Jones, 1984; Sparre and Venema, 1998). Third, current exploitation rates are compared to biological reference points estimated via relative yield-per-recruit models (Beverton and Holt, 1964). Finally, mesh-size increases and effort reductions are discussed as potential management measures for a more sustainable exploitation of Chwaka Bay's resources.

Chapter IV shifts the fisheries assessment conducted in Chapter III towards an ecosystem-based approach in order to answer RQ III. For this purpose, a trophic food-web model using *EwE* is constructed to describe the current trophic flow structure of the system. The trophic and network indicators together with the community energetics are used to evaluate the overall state of the Chwaka Bay ecosystem. The fishing pattern and the state of the fishery are characterized using the trophic level of the catch, CPUE, CPUA and the overall fish biomass. More importantly, the Chwaka Bay model is used to disentangle the impacts of the different gears in use on the ecosystem and its fish community. The Mixed Trophic Impact routine is used to evaluate the relative impact on functional groups and the ecosystem. Furthermore, the selectivity of each gear is characterized by the mean size of the catch and the catch composition diversity (Shannon Wiener Index). For the assessment of the relative impacts on the fishing community, the number of operating fishermen per gear is estimated and the overall profit as well as individual profit of each gear is calculated.

The steady state *Ecopath* model of Chapter IV is used in Chapter V to address RQ IV by simulating the effects of 1) a complete ban of the dragnet fishery, 2) a reallocation of dragnet fishermen proportionally to the other gears in use and 3) a combination of effort reduction and reallocation of dragnets.

Finally, the findings of this dissertation are summarized in Chapter VI. I first synthesize the results of Chapter III – V to define the status of the Chwaka Bay fishery. Secondly, I compare the findings of Chapter III with results of a participatory workshop conducted with 25 fishermen in September 2016 on resource status, gear impacts and potential management plans. I then use these findings together with a vulnerability analysis of the dominant target groups to identify 1) those families that show indication of overexploitation and 2) those families that possess the risk of being overfished due to their vulnerability to fishing and their importance for the fishery. Thirdly, I summarize the different ecological and socio-economic impacts of the different gears in use. Fourthly, I discuss potential mesh-size regulations, effort and gear control as well as spatio-temporal management measures in order to give final recommendations for future management directions. Fifthly, I discuss the current data collection system and suggest possible improvements as well as suitable biological and economic indicators to monitor Zanzibar's fisheries. Finally, I highlight the limitations of the applied methodology and provide suggestions for future research directions.

CHAPTER II - State of the Inshore Fisheries in Zanzibar

Chapter II.

State of the inshore fisheries in Zanzibar - synopsis

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Abstract

Most small-scale fishing communities are characterized through a high dependency on fisheries resources, a steadily increasing population density and the lack of alternative livelihoods. As a result, such fishing communities are highly vulnerable to the implementation of restrictive management measures. In this context, the evaluation of performance indicators of target resources and ecosystems together with proper management measures are imperative. In data-poor, small-scale fisheries, like those of the island of Zanzibar (Tanzania), the evaluation of the status of resources is often done using inadequate indicators (e.g. CPUE), which may eventually lead to strong management interventions. Mismanagement can lead to the distrust between fishermen and authorities, can weaken the compliance with management measures, and above all can increase food insecurity. In this paper we review the current literature 1) to search for and quantify the evidence for the overexploitation status of the fishery resources in Zanzibar; 2) to identify the most vulnerable fishing grounds and target species; and 3) to determine thresholds of sustainable fishing effort levels. Information was derived from fishermen`s perceptions, ecological surveys and from fisheries data. Results reveal that most studies have focused on the perception of fishermen and ecological inventories. However, fishermen's perception directly translates to individual catch rates, which are inappropriate to evaluate the status of resources, because they naturally decline as fishing effort increases. Ecological surveys on the west coast of Unguja Island suggest that current fishing effort and pattern has led to biomass reductions of many target species below the 50 % of the virgin biomass threshold. Similar indications were found for the collection of some invertebrate species at the north and east coast of Unguja. However, none of the studies directly quantified fishing effort level or fishing pattern, hindering the possibility to link overfishing indicators to fisheries input controls. In contrast to the findings from questionnaires and ecological survey, a catch-based analysis on officially reported landings indicates that none of the target families, except clupeioids show as yet signs of overfishing. Our review highlights that despite years of concern for an unsustainable resource use, we still know little about the general state of Zanzibar's fisheries resources.

Keywords: Zanzibar, overexploitation, fisheries management, artisanal fishery, indicators

2.1. Introduction

Small-scale fisheries worldwide employ over 90 % of capture fisher (FAO, 2015) and are the major livelihood and protein supplier in many coastal communities around the world (Allison and Ellis, 2001; Chuenpadgee, 2011). Particularly in the Western Indian Ocean fisheries resources can provide up to 70 % of animal protein and fisheries often employ more than 50 % of the local population (van der Elst, 2005; Wamsley et al., 2006; Jiddawi, 2012; McClanahan et al., 2013; Barnes-Mauthe et al., 2013). This high dependency under a steadily increasing population density and the lack of alternative livelihoods in most coastal communities underlines the prime importance of the management of small-scale fisheries (Drammeh, 2000; Jacquet et al., 2010; McClanahan et al., 2008b; Najmudeen and Sathiadhas, 2008; Nordlund et al., 2013; Walmsley et al., 2006). Furthermore, the high dependency makes coastal communities highly vulnerable to the implementation of restrictive management measures. Mismanagement leads to distrust between fisheries authorities and fishing communities and can weaken the compliance with management measures (Boonstra and Bach Dang, 2010).

The small-scale fisheries in the Western Indian Ocean are mainly multigear and multispecies fisheries that are carried out mostly in the near shore areas (Gell and Whittington, 2002; Jiddawi and Ohman, 2002; Mangi and Roberts, 2006). The use of conventional output control measures (e.g. quota systems) is highly challenging in such fisheries given the lack of financial and institutional capacity for monitoring and enforcing these measures on a multitude of species (Pomeroy, 2012; Salas et al., 2007). Therefore, small-scale fisheries managers highly rely on input measures such as gear-type and effort control as well as temporal and permanent closures of important habitats. Particularly, the control of gear-specific or overall effort and the permanent closure of fishing grounds are highly sensitive management actions, since effort reduction or gear restrictions can lead to the loss of livelihoods and ultimately to increased food insecurity (Cinner et al., 2011; Diegues, 2008; Salayo et al., 2008). Whether or not and to what extend such management measures are implemented strongly depends on the evaluation of the status of target resources and the respective ecosystem. However, to be able to evaluate the status of any resource or system there needs to be a reference level or baseline to compare with (Hall and Mainprize, 2004). Performance indicators enable managers to act accordingly, when thresholds are approached or exceeded and to evaluate the performance of subsequent management actions. Within the Western Indian Ocean region like in many other tropical coastal areas information on the dynamics of resources and the fishery is very scarce (van der Elst et al., 2005), making the formation of adequate management plans highly difficult.

Such an example is the fishery of Zanzibar, where marine fisheries resources form the basis for the livelihood of the majority of the inhabitants (Jiddawi and Ohman, 2002; January and Ngowi, 2010). Furthermore, the fishery sector in Zanzibar is one of the top three growth sectors (January and Ngowi, 2010). Yet it is still small-scale with 95 % of the fishery conducted within the near shore areas (Jiddawi and Kathib 2010). As the fishery of Zanzibar is open-access (Sobo 2004) with low fees for fishing licenses and no tax payment for artisanal fishermen, anyone can participate. Hence the number of fishing boats has more than doubled between 2003 and 2010 (Kathib and Jiddawi, 2010). The increase in fishing effort, the lack of control, together with the use of destructive gears and small mesh sizes are said to have led to an overexploitation of the fisheries resources. While, several authors state that the inshore fishery shows serious signs of overexploitation (Colbert-Sangree, 2012; de la Torre-Castro and Lindström, 2010; Francis and Bryceson, 2001; Jiddawi and Ohman, 2002; Mkenda and Folmer, 2001; Ngusaru et al., 2001; Payet et al., 2001; Phelan and Stewart, 2008; Silva, 2006; Thyresson et al., 2013), others refer to specific fisheries resources (Eriksson et al., 2010; Fröcklin et al., 2014; Thyresson et al., 2011; Torell et al., 2007) or specific areas (Colbert-Sangree, 2012; Jiddawi, 2012). The lack of information on fishing effort and on the dynamics of species-specific catches makes it highly difficult to verify these concerns. Nevertheless, if Zanzibar's fisheries management ought to be successful an understanding of the status of its resources and the identification of sustainable levels of fishing effort is imperative.

In this paper we review the current literature 1) to quantify the extent of overfishing in Zanzibar; and 2) to identify the most vulnerable fishing grounds and target species. We further use the catch-based method (Froese and Kesner-Reyes, 2002) to classify the fisheries of the reported target groups into *developed*, *fully exploited*, *overfished* and *collapsed*.

2.2. Methodology

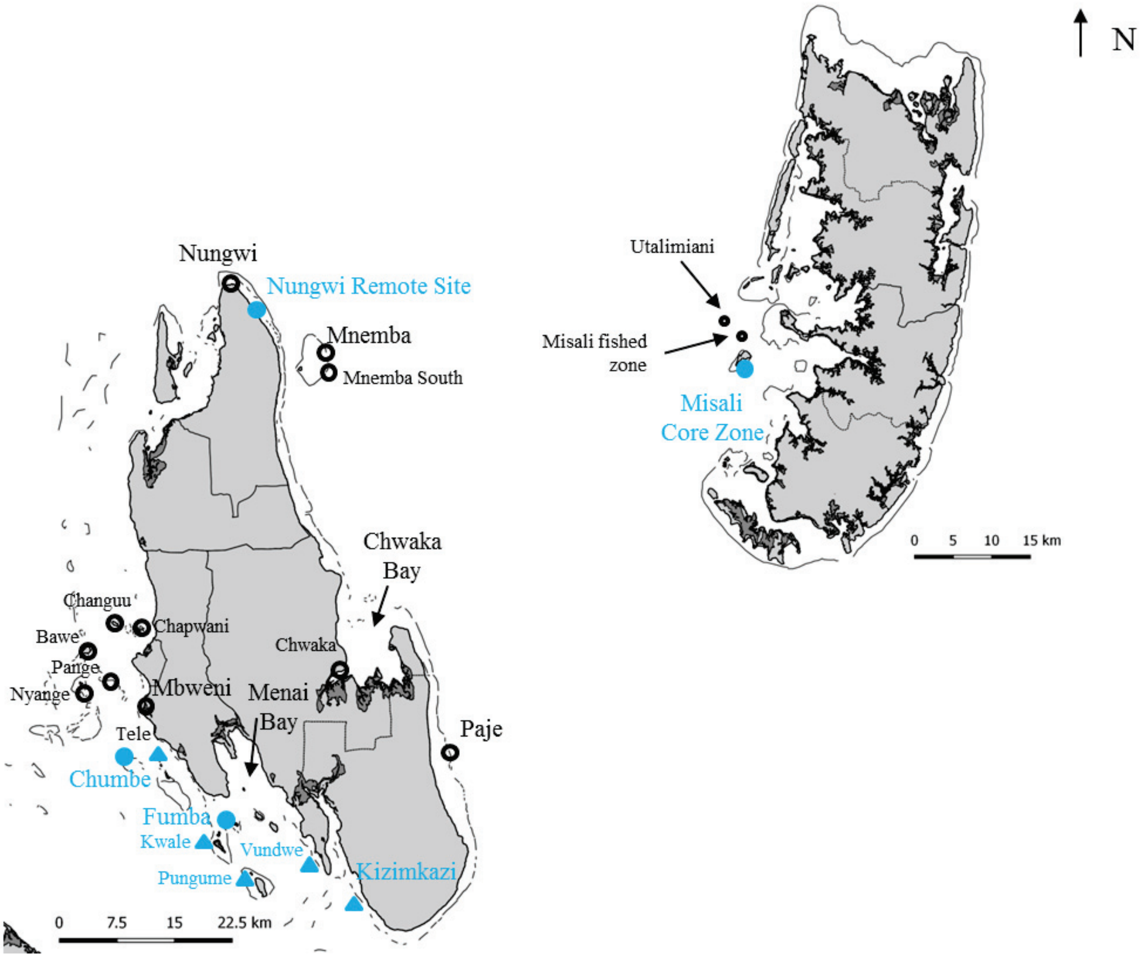

Fig. 2.1. Map of Unguja and Pemba. Depicted are the different study sites of the ecological surveys. Blue circles indicate fully protected sites and blue triangles indicate partially protected sites.

2.2.1. Study area and literature review

Zanzibar is a semi-autonomous island state in the Western Indian Ocean and belongs to the United Republic of Tanzania. Zanzibar consists of two major Islands Pemba and Unguja (Fig. 2.1.), which lay 40 - 60 km off the mainland. Total fishing area of Pemba and Unguja has been estimated at roughly 4001 km^2, with an estimated number of 27,187 fishermen (approx. 7 fisher km^{-2}). The fishery is highly artisanal with traps, seines nets, handlines and spears being the most dominant gears. Zanzibar is divided into 10 districts with a total of 224 landing sites (Kathib and Jiddawi, 2010).

We searched the existing literature for evaluating the status of Zanzibar's nearshore fisheries using the key word "Zanzibar" in combination with either "fishing", "fishery", "overexploitation", "overharvesting", "overfishing" or "resource extraction" in the Web of Science and Google Scholar. Studies that were designed to assess the impacts or the status of Zanzibar's fisheries as well as reports and reviews about the fishery, were reviewed in detail and are presented in the following sections. We divided these studies into 4 groups: (1) Questionnaires and interviews with fishermen; (2) Ecological surveys; (3) Invertebrate studies; (4) Fisheries statistics and stock assessments. The different sites in Zanzibar investigated by each reviewed study are presented in Fig. 2.1.

2.2.2. Defining thresholds of overexploitation

We define overfishing as the level of fishing effort that reduces stock levels below safe biological or sustainable limits. We further use the terms overfishing and overexploitation synonymously. Sustainable fishing is defined as the level of fishing that can be maintained indefinitely. In conventional fisheries assessments quantifiable measures are usually linked to single species models that describe the population dynamics of resources (e.g. per recruit models, surplus production models, dynamic pool models; Hall and Mainprize, 2004). One widely applied reference point is B_{MSY}, which is the level of biomass that produces maximum sustainable yield. However, such measures do not take into account fisheries effects on the biomass of non-target species, the habitat, or the structure and function of the ecosystem. In this context the term overfishing and the definition of tangible reference points becomes much more complex and indicators need to go beyond biomass estimations for target groups. This review does not aim at defining ecosystem overfishing nor do we aim at identifying ecosystem-based indicators. Ecosystem overfishing has been characterized by Murawski (2000) and several authors have provided measurable reference points to assess the level of ecosystem overfishing (Coll et al., 2008; Link, 2005; Rochet and Trenkel, 2003). Link

(2005) examined a suit of proposed indicators and offers several quantifiable measures together with some gross limits for these measures. However, to shed light on the question whether or not Zanzibar's nearshore fisheries can be classified as overexploited, we applied the following crude warning thresholds to the data collected by several different ecological surveys throughout Zanzibar: 1) Stock is reduced to half of the unexploited population size of target and non-target species (B_{50}/N_{50}, Gulland, 1973); 2) Species richness is reduced significantly (beyond 10 % , SR_{90}); 3) The target species/group shows a 30 % reduction in its mean length (L_{70}, Link et al., 2005). We only considered changes below thresholds to be a warning sign to overfishing, when observed changes were significant ($p = 0.05$). In cases were the authors did not provide information about whether or not findings were significant, we used the mean values together with the standard deviation and the number of samples to test for significance with a one-tailed t-test.

2.2.3. Analysis of Zanzibar's annual catches

The Department of Marine Fisheries Resources (DMFR) collects monthly data on the landings (kg) of 19 target groups (Siganidae, Scaridae, Lethrinidae, Serranidae, Mullidae, Lutjanidae, Mugilidae, clupeioids, sardines, mackerels, Carangidae, tuna-like fishes, marlins & sailfishes, kingfish, Sphyraenidae, sharks and rays, molluscs, lobsters and other demersal & pelagic fish). Data collection takes place at 31 landing sites distributed over the different districts. Monthly data of these landing sites is extrapolated to Zanzibar using fixed raising factors. The monthly landing data is available on hard copies and stepwise aggregated. The final aggregated stage comprises the annual landing data of the 19 groups per district and is available in soft copy (Kathib and Jiddawi, 2010). Zanzibar's annual landing data of the 19 groups is then reported to the FAO, which pool kingfish and mackerels together into the group seerfishes nei.

We used the catch-based method proposed by Froese and Kesner-Reyes (2002) to classify the fisheries for the different target groups (excluding other demersal & pelagic fish) into *developed*, *fully exploited*, *overfished* and *collapsed*. For this purpose the time series of catches is divided into 2 periods: before and after the year of historical maximum catch (C_{max}). In the period before C_{max} the catches are classified into developed ($< 0.5\ C_{max}$) and fully exploited ($\geq 0.5\ C_{max}$). In the period after C_{max} the catches are classified into *fully exploited* ($\geq 0.5\ C_{max}$), overexploited ($0.1 – 0.5\ C_{max}$), or collapsed ($< 0.1\ C_{max}$). However, since the catch-based method has been criticised for overly classifying fisheries as overexploited or collapsed due to stochasticity (Branch et al., 2011), we first applied a *Loess* smoothing to the raw data as proposed by Anderson et al (2012) before classifying the catch time series into the 4 classes (See an example of the *Loess* smoothing in Fig. S.2.1).

Two data sets were obtained in 2014 from the DMFR 1) total landings for Zanzibar covering the period from 1990 – 2012 and 2) landings for Unguja Island covering the period from 1990 – 2010 (raw data is provided in Table S.2.1.). The first data set was complemented with information for 2013 and 2014 obtained from the official fisheries production statistics provided by the FAO. For mackerels and king fish we only used the data from DMFR (until 2012), due to inconsistencies found in the FAO data. The catch-based method was applied to the data set for Zanzibar and for Unguja separately.

2.3. State of the fishery

2.3.1. Interviews with fishermen

One of the central observations made about the fishery on the island is that fishermen repeatedly report declines in their catch rates sometimes with respect to a certain target resource (Table 2.1.). For instance semi-structured interviews conducted in 2002/2003 and again in 2014 with fishermen from Chwaka Bay (Fig. 2.1.) revealed a perception of a decrease in individual catches (de la Torre-Castro and Rönnbäck, 2004; Geere, 2014). Particularly, the catches of Carangidae, Lutjanidae and Serranidae were said to experience a constant decline (de la Torre-Castro and Rönnbäck 2004). Two of the central reasons given by the fishermen for the decline in fish catch, was the use of dragnets (70 %) and the increase in the number of fishermen (22 %). The same observation of declining catch rates was reported by Colbert-Sangree and Suter (2015) for fishermen from Kizimkazi Dimbani and Jambiani (Menai Bay, Fig.2.1.). The authors found that respondents would generally report a decrease in their average weekly catches over the last 20 years. Furthermore, the respondents stated that catches had been 188.60 % (Kizimkazi Dambiani) and 97.11 % (Jambiani) higher than today. Fishermen perceived that one of the biggest problems in both villages is the continuous use of illegal fishing methods. However, contrasting to Chwaka Bay many respondents named a decline in fish stocks as the major reason for the decline in catch rates, rather than an increase in fishing effort. The perception of declines in catch rates has also been observed in a study conducted in 2011 to assess the migration pattern of fishermen from Pemba (Wanyonyi et al. 2016, Fig. 2.1.). The reasons for fishermen to migrate to other fishing grounds are divers and complex. However, 60 % of the fishermen questioned by the authors would mention that a decrease in the availability of fish within their own fishing grounds has let them to migrate to other places. Besides the general report of decreasing catch rates, there are also observations about decreasing catches for specific resources such as parrotfish (Thyresson et al., 2011) or sardines (Stanek, 2015). Stanek (2015) asked fishermen involved in the sardine fishery of Mwangapwani (Fig. 2.1.) about their perception on the status of their fishery, of which 60 % reported declines in catches and several respondents mentioned a large increase in the number of fishermen as the core problem.

The methodology of conducting questionnaires in order to assess the decrease or increase of a particular resource is common if no other information is available. However, the potential bias in the perception of interviewees, which limits the power of individuals to perceive trends, has been discussed by several different authors (Daw, 2010; Daw et al., 2011; Papworth et al., 2009; Verweij et al., 2010). These studies indicate the importance of cross validating findings conducted by questionnaires with statistical analysis of catch and effort trends. Fishermen rely on relatively short-term experience, when assessing the status of their resources, which can impede their power to assess long-term trends in resource abundance (Ainsworth and Pitcher, 2005). Additional to these temporal limitations in their ability to judge on the status of fish stocks, fishermen have often only an overview of parts of their target populations, because of the selectivity of their gears and mesh sizes and their spatial restrictions. Thus, questionnaires need to be designed in a way that 1) the subsample of interviewees represents the different types of gears used to catch the resource of interest and 2) that the interviewed fisher fish in the total home range of that resource. In addition, studies have shown that fishermen are likely to be more pessimistic about the status of their target resources than stock assessments would suggest (Ainsworth and Pitcher, 2005; Rochet et al., 2008). Papworth et al. (2009) also points out that fishermen, in particular older ones, may remember past resource conditions wrongly, which is called "memory illusion". These memory illusions might be present, when perceived trends are exaggerated because of occasional very large catches in the past. More importantly, the perception of fishermen of declining resources directly translates into catch per unit of fishermen. However, in the case of Zanzibar it has been shown that there is a steady increase in effort (Kathib and Jiddawi, 2010; Tobey and Torell, 2006). If more fishermen are targeting the same resources there will be an increased removal and as such it is evident that catch rates will decrease even though fish stocks might not be overfished yet (Kolding et al., 2014). This becomes also evident, when looking at the responses of most fishermen, who argue that one of the central problems in the decrease of their catches is the increase of participating fishermen. Moreover, in fisheries science theory, yield is maximized, when the abundance level of a stock approaches half the unexploited population size (B_{50}, Maunder, 2006). Consequently, fishermen will almost always experience a decrease in their catch rates, when effort increases, but only if the stock size is reduced beyond B_{50} there is a situation of overfishing (Kolding et al., 2014; Maunder et al., 2006). Lastly, questionnaires with fishermen about the status of resources will not help in setting target limits or generate any quantitative management measures. The complaints of fishermen about decreasing catches and subsequent income, however, should generate concern for food security and the sustainability of fishing as a livelihood.

Table 2.1. Overview of the perception of fishermen about the status of fisheries target resources at different sites throughout Zanzibar.

	Fishermen's perception					
	Overview		**Specific sites**			
Type of Resource	Unguja	Pemba	Chwaka Bay	Menai Bay	Nungwi	Mwangapwani
General nearshore resources	-	Overfished	Overfished	Overfished	-	-
Carangidae	-	-	Overfished	-	-	-
Small pelagics	-	-	-	-	-	Overfished
Serranidae	-	-	Overfished	-	-	-
Lutjanidae	-	-	Overfished	-	-	-
Scaridae	Overfished	-	-	-	-	-
Invertebrates	-	-	Overfished	-	Overfished	-
Sea cucumber	Overfished	-	-	-	-	-

2.3.2. Ecological surveys

Ecological surveys, such as underwater visual census surveys, can aid in the assessment of fishing effects that go beyond the reduction in the biomass of target species. Often such studies measure a variety of different indicators such as benthic cover to assess impacts of fishing on habitats, changes in species diversity to assess the loss of species and the reduction of biomass of non-target species. However, without clear thresholds or reference points such assessments remain at the level of simple inventories. The condition of indicators in the past or in unfished sites can provide helpful baselines. On Zanzibar most of the ecological surveys (i.e. underwater visual census) conducted to assess the impacts of different levels of fishing intensities used unprotected or partially protected sites as comparative source. An intensively studied fishing area on Zanzibar is the west coast of Unguja (Fig. 2.1.), where several small coral reef islands and sandbanks attract tourists and fishermen. This area is highly suitable to study the impacts of fishing, because part of one of these reef islands is a well-enforced marine park since 1991 (i.e. Chumbe Island Coral Park) and thus serves as a potential baseline to assess changes in different ecological indicators.

McClanahan et al. (1999), for instance, investigated the effects of fishing at southern Kenyan and northern Tanzanian coral reefs including the protected Chumbe Island Coral Park and the two unprotected reefs Changuu and Chapwani (west coast Unguja, Fig. 2.1.). Generally the authors found a significantly higher overall fish density and for several target fish families a significantly higher biomass in the protected sites compared to the unprotected sites. When comparing the biomass, reported by the authors, of Chumbe, Changuu and Chapwani (Table 2.3a.), the latter two showed significantly depleted biomass values below B_{50} in 6 and 8 out of 10 fish families. Species richness in the two unprotected sites were significantly reduced below warning thresholds (< SR_{90}) for Acanthuridae and Scaridae. Lokrantz et al. (2010) studied the same reefs together with Bawe, Pange and Nyange (Fig.2.1.) at the west coast of Zanzibar in 2004 and 2006. In their study, the authors compared the abundance, species richness, species diversity and size-class distribution of herbivorous fish in order to detect impacts of different levels of fishing on the herbivorous fish community. Fishing intensity was based on 1) number of days per week spent fishing per fishermen, 2) the total number of fishing vessels or households at the different landing sites and 3) distance of landing site to reef. The authors found a decreasing gradient of fishing intensity from North to South, with Chumbe being the reef with zero fishing intensity. Results confirm those found by McClanahan (1999), indicating that there has been no improvement for the herbivorous fish community at Changuu. Almost all measured proxies for the status of herbivorous reef fish (abundance, species richness, species diversity and biomass) were negatively correlated with reef-specific fishing pressure. When examining the reported biomass values of herbivorous fish (excavators, grazers and scrapers), it appears that biomasses at all unprotected sites were significantly decreased below B_{50} (Table 2.3a.). While Pange, Bawe and Changuu show a reduction in species richness of 45 to 70 %, in Nyange excavator species richness did not exceed the warning threshold (SR_{90}). In addition, the authors showed that the average length of excavators was also significantly reduced below L_{70} in Pange, Bawe and Changuu (Table 2.3a.).

Contrasting to the findings of McClanahan et al. (1999) and Lokrantz et al. (2010) a study conducted by Tyler et al. (2011) could not find any significant difference in abundance, biomass or mean length of fish within unprotected sites (Changuu, Bawe, Pange, Chawacha and Paje) compared to 5 partially protected reef sites[3] within the Menai Bay Conservation Area (Fig. 2.1., Table 2.3c.). Furthermore, the authors could not find any significant difference in important habitat variables such as live hard coral cover between protected and unprotected sites, indicating that the level of destructive fishing on the unprotected reefs does not significantly affect the habitat. However, restricting the use of illegal fishing methods within the Menai Bay Conservation Area has significantly increased species richness of nine commercial fish families and the unprotected reef shows a reduction in species richness below warning thresholds ($< SR_{90}$, Table 2.3c.). An earlier study by Tyler et al. (2009) also investigated the differences in species richness of fish at different depths in fully protected (Chumbe, Misali core zone and Mnemba, Fig.2.1.) and 1 to 2 unprotected reefs close by. The authors argue that the most common fishing methods on the island are restricted to shallower areas. Correspondingly, the authors found that richness of commercial species was significantly depleted only in shallow areas ($< SR_{90}$, Table 2.3c.), but not in deeper areas of the fished sites compared to the protected sites.

Similar to the study by Lokrantz et al. (2010) a recent study by Aller et al. (2014) investigated the impact of different levels of fishing on the density, species richness and species diversity of fish, but in seagrass beds than coral reefs. Chosen sites were located throughout Zanzibar (i.e. Changuu, Mbweni, Chumbe, Fumba, Chwaka and Nungwi, Fig. 2.1.). To assess the level of fishing on the different sites, the authors used the number of houses within a 3 km^2 radius as a proxy for fishing effort[4]. Overall their findings were that the level of development was negatively related to fish density,

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[3] Prohibiting the use of illegal fishing methods (e.g. dynamite, dragnets, poison).

[4] assuming that fishermen usually travel 3 km to their fishing grounds, which is based on reports from UNEP, 2011 and Lokrantz, 2010.

species richness and species diversity. Furthermore, authors found that the level of development had the strongest effect on fish assemblage structure considering other local and regional factors such as wave exposure, depth and seagrass characteristics. Authors did not report the estimated species richness and diversity of sampled sites, we thus could only compare mean fish densities between sites of zero fishing intensity (mean densities of Nungwi and Fumba) and fished sites (Chumbe[5], Mbweni and Changuu). We found no significant differences for mean fish densities between the unprotected and protected sites, except for one functional group at Mbweni (invertebrate feeder fish), which showed density depletions below thresholds (N_{50}, Table 2.3a.).

Finally, Daniels et al. (2003) investigated the effects of fishing on Pemba using 2 fished sites and 2 protected sites around Misali Island. The authors found significant differences in the abundance of 5 fish families. However, of these families 3 had a higher abundance in the fished site compared to the protected site. Only the biomasses of Holocentridae and Scaridae were 53 to 70 % lower in the extraction zone ($< N_{50}$, Table 2.3b.). Furthermore, the authors could not find a significant difference between the mean lengths of fish in the fished and protected site. However, the lack of detecting any effect related to the reduced fishing intensity might be due to the relatively small area (1.4 km^2) of the protected site and the fact that it was only implemented three years before the study was conducted.

Results of the above-mentioned studies indicate that the reefs on the western side of the islands are subject to a very intensive fishing pressure. Using Chumbe Island as a reference site the estimated species richness and biomass values of fish in Changuu, Bawe and Pange exceed warning thresholds and should trigger further investigation. While, Nyange reefs seem to be less impacted, their current fishing effort could be treated as a level to which the effort in Changuu, Pange and Bawe should be reduced. Since Lokrantz et al. (2010) only provide approximations of fishing intensity, further fisheries surveys are needed that quantify fishing effort in order to set appropriate reference points.

[5] Only part of the Chumbe Island is under protection. The sites chosen by the authors were located within the part that is still open to fishing.

2.3.3. Invertebrate fisheries

Invertebrates are harvested by gleaning in intertidal areas an activity that is mainly conducted by women and children (Nordlund et al., 2010). They represent a very important source for food security of local households, as a large part of the harvest is for home consumption. However, studies have shown that the importance of invertebrate collection as a source of income is steadily increasing (Fröcklin et al., 2014) and in areas such as Chwaka Bay on the east coast of Zanzibar this type of fishery contributes significantly to the household's income. Moreover, invertebrates comprise the largest proportion of marine export products of Zanzibar (Jiddawi and Ohman, 2002). For instance, sea cucumbers, known as "beche de mer", are only used for export (Eriksson et al., 2010).

A study by Nordlund et al. (2010) conducted in 2007 and 2008, investigated the social and ecological effects of invertebrate harvesting in the northern part of Zanzibar. Authors conducted interviews with 18 female invertebrate collectors from Nungwi village and they compared abundance and species richness of epibenthic invertebrates in an exploited site (Nungwi village, Fig. 2.1.) with a remote site. Similar to other studies about fishermen's perception on resources status, 94 % of interviewees observed a decline in the number of invertebrates in the last 5 - 10 years (Table 2.1.) and one of the most frequently mentioned reason was the increase in the number of invertebrate harvesters. The inventories revealed a significantly higher species richness and abundance of epibenthic invertebrates at the remote site compared to the exploited site ($< SR_{90}$; $< N_{50}$, Table 2.3b.). Crustaceans and sea cucumbers showed a reduction in mean biomass within the exploited site below warning thresholds ($< N_{50}$). However, since authors do not provide information on standard deviation of these mean values, it is not clear whether or not the observed differences for crustaceans and sea cucumber abundance are significant.

Similar findings to that of Nordlund et al. (2010) were found for the Chwaka Bay invertebrate fishery. Fröcklin et al. (2014) conducted semi-structured interviews in 2005 and 2010 with invertebrate collectors from the bay and conducted biological inventories on 11 important fishing grounds. Authors complemented their study by comparing the catches of invertebrate collectors in 2005 and 2010. For that, they analysed the catches of 10 female invertebrate collectors over a period of 14 days in both years as well as the catches of 22 (2005) and 23 (2010) male invertebrate collectors. The perceived decrease in resource abundance (88 % of respondents, Table 2.1.) was supported by the findings from the biological inventories, which showed that gastropod and bivalve abundance had decreased below N_{50} (Table 2.3b.). Particularly, the high value species Gold ringer (*Cypraea annulus*), Philipine horse mussel (*Modiolus* *philippinarum*), Pen shell (*Atrina vexillum*) and the Humpbacked conch (*Gibberulus gibberulus*) showed significant declines. Catches of high value species of lobsters and sea cucumbers were also found to be rare in both years and were also absent from the inventories.

The paucity of commercially important species of sea cucumber are in accordance with findings from another multidisciplinary study on the status of the sea cucumber fishery on Zanzibar (Eriksson et al., 2010). In 2007, Eriksson et al. (2010) conducted interviews in nine villages around the island with inter alia fisher and middlemen (Fig. 2.1.), revealing that 94 % of the fishermen and 92 % of the middlemen said it was harder to find most species nowadays than before (Table 2.1.). In addition, in 2009 the authors monitored the catch and effort of sea cucumber collectors in three villages (Mkokotoni, Fumba and Uroa, Fig. 2.1.) over a period of two weeks and took length measurements of the highly valuable *Holothuria scabra*. Authors also conducted a visual census of sea cucumbers at Ukumbe, Kwale and Chumbe (protected site) and compared species diversity and density. The catch analysis revealed that between 70 % and 95 % of the catch composition was comprised of low value species. Based on a length at maturity of 160 mm, the authors estimated that 67 % of the catch of *Holothuria scabra* consisted of immature individuals. In addition, their visual assessment of the sea cucumber population revealed a tenfold higher density of medium value species in Chumbe than in the unprotected site (< N_{50}). The authors also found a higher species diversity of sea cucumber in Chumbe compared to Kwale and Ukombe (< SD_{90}, Table 2.3b.). Eriksson et al (2010) and a later study conducted by Eriksson et al. (2012) also found that the sea cucumber fishery is not only taking place in the nearshore areas of Zanzibar, but that scuba divers target offshore areas leaving almost no protection for this invertebrate group. Analyses indicate that the scuba diving fishery seems to be resistant to the declines in the sea cucumber populations and imposes more pressure on the resources through diversification of their catches.

The findings of the above-mentioned studies indicate that several gastropod, bivalve and sea cucumber species are harvested at a rate that exceeds warning thresholds. Factors such as high accessibility, slow growth and limited immotility make many invertebrate species highly vulnerable to overexploitation (Jamieson, 1993; Perry et al., 1999). This highlights the need to assess the magnitude of invertebrate gleaning and the level of current exploitation of the main target species of the fishery.

2.3.4. Fisheries assessments and catch trends

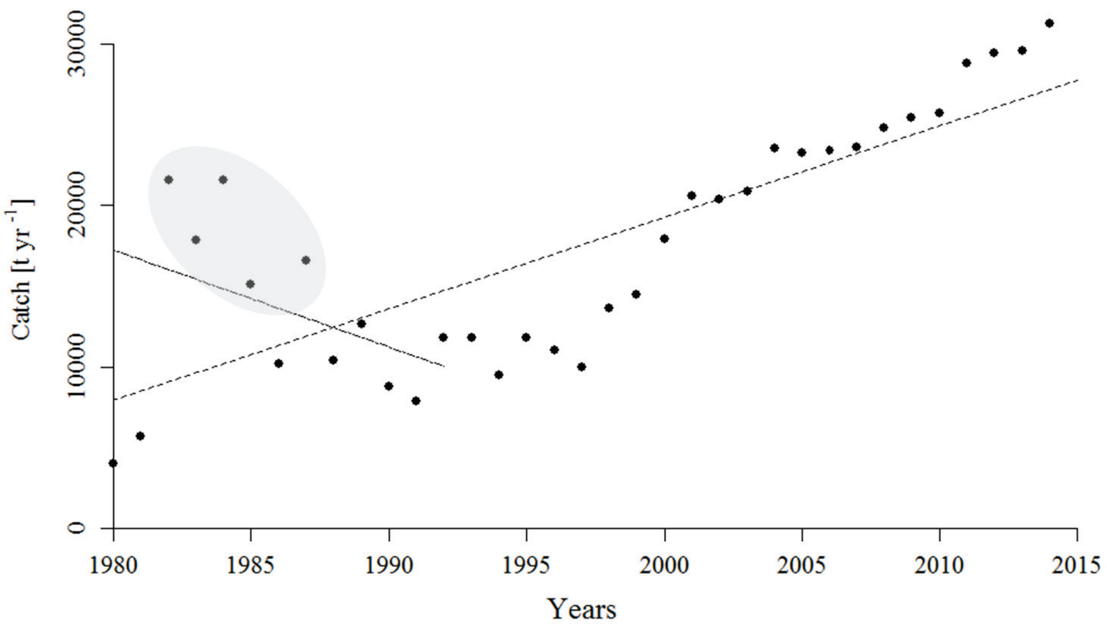

Fig. 2.2. Zanzibar's total annual catch [t] between 1980 and 2015. Marked in grey are the years that show an unusual high catch. Isolated catches between 1980 and 1994 show a declining trend, while the whole time period between 1980 and 2015 reflect an overall increasing trend.

Annual statistics of Zanzibar's total landings and fishery specific landings showed declining trends between 1980 and 2000. For instance, Jiddawi and Ohman (2002) reported that the annual catch on Zanzibar has declined from 20,000 t in the 1980's to approximately 15,000 t in 2000. Similarly, the catches of small pelagics from boats of the Zanzibar Fisheries Corporation dropped from 600 t in 1986 to 91 t in 1997 and the landings of the reef fishery declined from approx. 3660 t yr^{-1} in 1990 to approx. 3450 t yr^{-1} in 1997. In addition, the exports of lobsters had declined from 23 t in 1993 to 0.7 t in 1997 (Jiddawi and Ohman, 2002). The government, however, believed that the decline in catches was a symptom of poor and inadequate fishing gear rather than an unsustainable fishery and promoted the development of the fisheries sector by that time (Mkenda and Folmer, 2001).

To address the question whether or not Zanzibar's nearshore fisheries provide room for expansion, Mkenda and Folmer (2001) used the official landings data to estimate the aggregated maximum sustainable yield of Zanzibar's nearshore resources and compared the resulting estimates with current catch and effort level. Analyses were based on Schaefer and Fox surplus production models using catch (t month^{-1}) and effort (fishing days fisher^{-1} month^{-1}) data of four different types of gears from 1980 to 1996.

The output of their analysis provided an annual maximum sustainable yield of 24,481 t y^{-1}, which was in line with rough estimations provided by the FAO (25,000 – 30,000 t yr^{-1}, FAO 1991). However, analysis also showed that the annual yield between 1990 and 1996 was only about 40 % of the calculated maximum sustainable yield. Concurrently, the average annual effort in that period exceeded with 657,762 units the calculated optimum effort of 361,446 units. Considering that the effort at that time was well above the effort needed for reaching the maximum sustainable yield, the authors argued that the nearshore stocks were biologically overfished and advised to not expand the fishery but to set transferable quotas on the basis of historical catches. A contrasting picture emerges when looking at the official annual landings of Zanzibar between 1980 and 2014 (Jiddawi, 2001; DMFR, 2014, FAO, 2017, Fig. 2.2.). Total annual landings increased from about 3900 t in 1980 to 31,267 t in 2014. Catches from the years 1982 to 1984 and 1987 are highly irregular comparing to the overall trend of landings. Thus, when considering only the landings prior 2000, particularly until 1991, Zanzibar's annual landings seem to decrease, though since 2000 landings are steadily increasing. The average monthly effort between 1980 and 1989 (14,837 fishing days $fisher^{-1}$) was only 30 % of the average monthly effort between 1990 and 2000 (49,661 fishing days $fisher^{-1}$, Mkenda and Folmer, 2001). Hence the high catches in 1982-1984 and 1987 cannot be explained by an increase in fishing effort.

In the absence of annual fishing effort the surplus production model could not be updated. Instead of looking at the development of Zanzibar's total catch, we used the catch-based method to evaluate the changes in the catches of 18 target groups. Results indicate that 10 (55.5 %) of the reported target groups reached maximum catch only in the last year of the time series and thus may be still developing (e.g. tuna-like fisheries, Fig. 2.3.), while 7 (39 %) were fully exploited in the last year of the time series (e.g. Siganidae, Fig. 2.3.). Only the catches of clupeioids were classified as overexploited since 2013 and none of the groups were classified as collapsed. There has been a perception of an under exploitation of the wider EEZ and the subsequent pelagic resources (Feidi, 2005), which triggered projects aiming at reallocating nearshore fishermen towards offshore areas (Gustavsson et al., 2014). However, our analysis of the overall catches suggests that five out of the reported eight pelagic groups have already reached a fully exploited or overexploited (clupeioids) status.

The officially reported landings analysed above are comprised of information from Pemba and Unguja. Since the fishery of Unguja is more developed than Pemba's fishery (Kathib and Jiddawi, 2010), we also analysed total annual catches of Unguja for the same target groups between 1990 and 2010. Similar to the overall trend many target groups (61 %) reached their maximum catch only in the last year of the time series (e.g. tuna-like fishes, Fig. 2.4.). Furthermore, none of the target group's current catch (2010) was classified as overfished leaving 39 % of target groups fully exploited (e.g. Siganidae, Fig. 2.4.). However, catches of lobsters were classified as overfished for the first time in 1992 and shortly after (1998) catch levels fell below 0.1 C_{max} (collapsed).

An exception to the overall increase in catches is the fishery of Chwaka Bay. Jiddawi and Ohman (2002) reported declining trends in the catch of small and large pelagics as well as reef fish for Chwaka Bay and more recent estimates by Jiddawi (2012) show that the total annual catch has continued to decline from 950 t in 1990 to around 370 t in 2003 and 310 t in 2007.

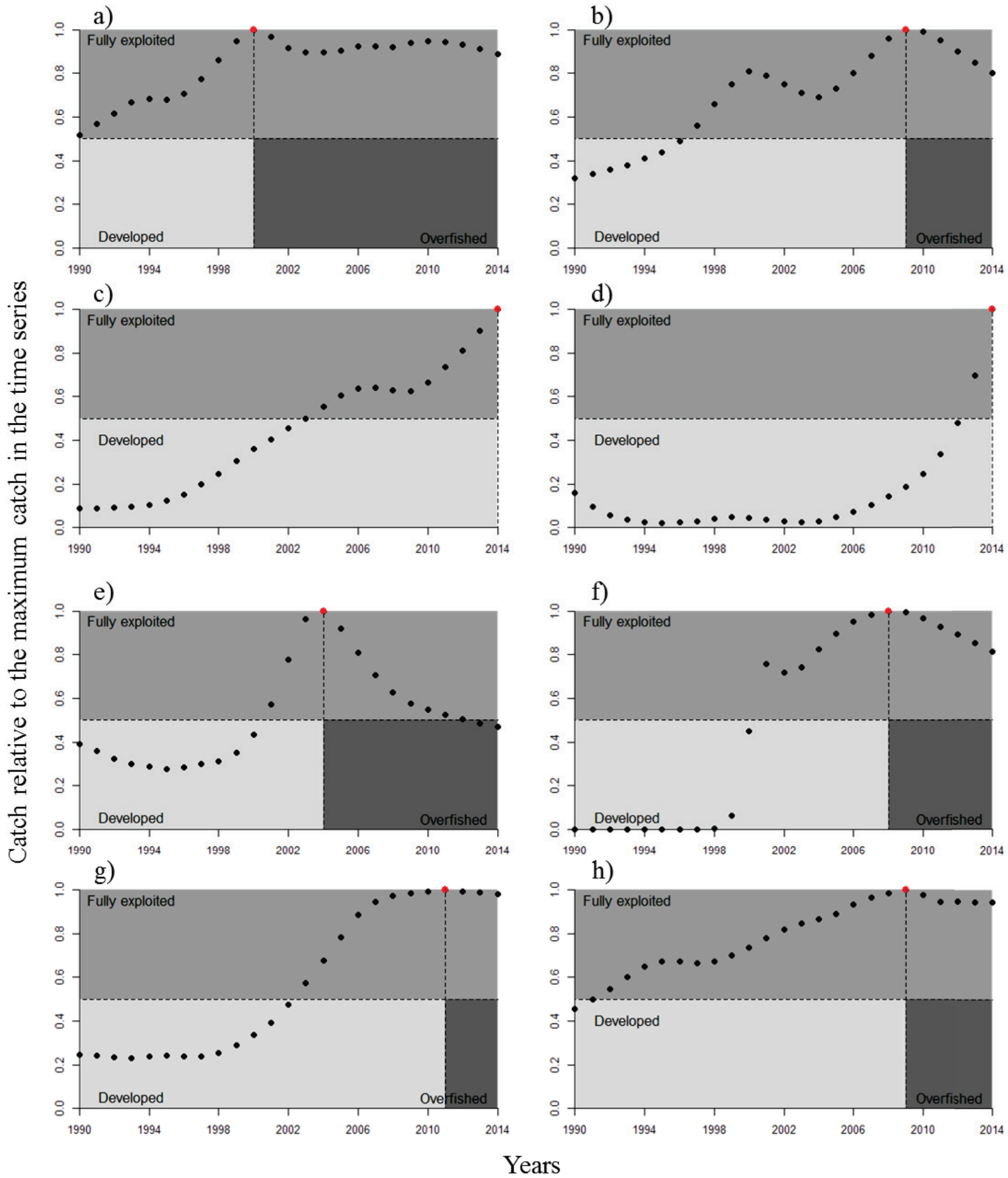

Fig. 2.3. Zanzibar's annual catch relative to the maximum catch (highlighted with red dots) of a) Siganidae, b) Lethrinidae, c) tuna-like fishes, d) lobsters, e) clupeoids, f) sardines, g) Carangidae, h) molluscs. The logarithm of the annual catch was smoothed with a LOESS function prior to classification.

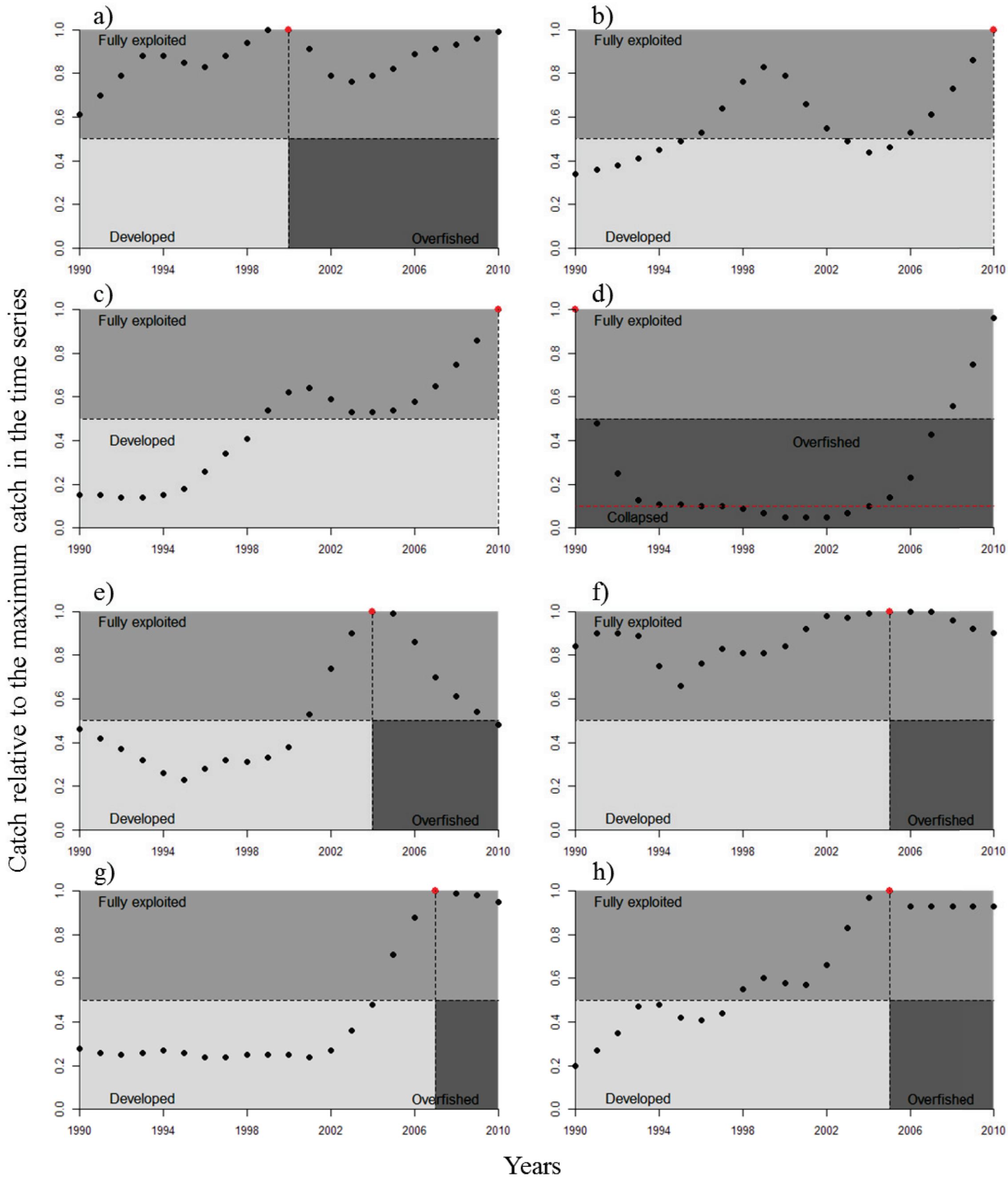

Fig. 2.4. Unguja's annual catch relative to the maximum catch (highlighted with red dots) of a) Siganidae, b) Lethrinidae, c) tuna-like fishes d) lobsters, e) clupeiods, f) mackerels, g) Carangidae, h) sharks and rays. The logarithm of the annual catch was smoothed with a LOESS function prior to classification.

2.4. Conclusion – The status of Zanzibar's resources

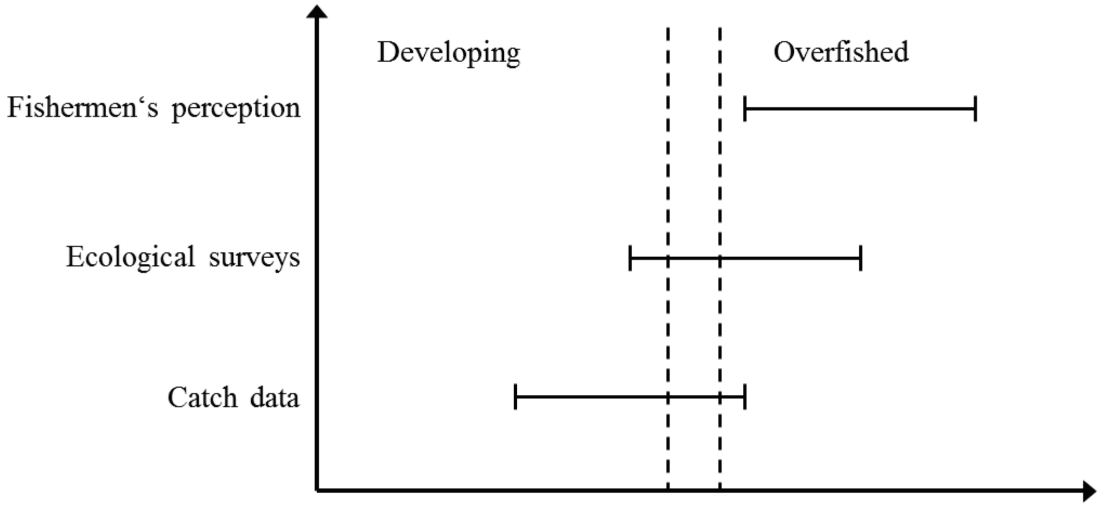

Fig. 2.5. Rough classification of the overall state of the inshore fishery of Zanzibar according to fishermen's perception, ecological surveys and catch data.

The findings of this review demonstrate that while fishermen's perception and ecological surveys point towards an overfished state of the Zanzibar's fisheries resources, the official reported catch data suggest an under- to fully exploited state (Fig. 2.5.). Although individual catch declines are not a sign of overfishing, for most of the rural communities fishing is the prime livelihood and hence such reports should trigger proper fisheries assessments including the investigation of different gear impacts in order to set appropriate management plans that allow for sustainable and profitable catches.

In fact, to draw conclusions of the general state of Zanzibar's nearshore fisheries, the only comprehensive information available is the annual catch of target groups. However, results of the catch-based method indicate that catches of most of the groups on Zanzibar as well as Unguja Island (55.5 %, 61 %, respectively) are still increasing. For several target groups (Siganidae, Scaridae, Serranidae, Mullidae and Lutjanidae) these findings, contrast fishermen's perception and findings from ecological surveys reporting local biomass/abundance depletion (Table 2.2.). Furthermore, it seems that several of the pelagic fish groups are more impacted than the demersal reef-associated groups. Analyses of Zanzibar's annual fisheries statistics reveal that the catches of four out of eight pelagic fish groups were classified as fully exploited in the last year of the time series. Small pelagics seemed to be particularly impacted since the catches of clupeiods are currently (2014) classified as overfished and sardines are perceived as overfished by fishermen from Mwangapwani. This, however, contrasts the general belief that Zanzibar's nearshore fisheries show serious signs of overexploitation and that pelagic fish are underexploited (Feidi et al. 2005). Nevertheless, catch data per target group is only available since 1990, though particularly reef fish have been exploited since long time to provide food and income to the local communities and some of the groups may have already reached their maximum catches before 1990. In this case our results would be biased by a shift in baseline. In addition, we are not able to judge whether or not increases in landings are partly explained by increasing reports of catches, enhanced data collection or a geographical expansion of the fishery. The picture that these aggregated statistics paint needs to be cautiously looked at, since information is provided on group and not species level and therefore mask the dynamics of individual stocks. Thus, the results could create a false perception of sustainability of Zanzibar's nearshore resources. The lack of annual or biannual information of effort makes it difficult to apply even less data requiring fisheries assessment methods such as holistic models like the surplus production models used in the study of Mkenda and Folmer (2001). Since 2000 only three fisheries frame surveys were conducted, which provide information on fishing effort (in 2003, 2007 and 2010). As such, there is a need to strengthen the capacity of the DMFR to estimate annual fishing effort and to collect information on annual catch at species level for some of the main target species.

Table 2.2. Overview of the status of Unguja's fisheries target groups according to the catch-based method in comparison to target groups that a) have been perceived as overfished and b) showed local reductions in biomass/abundance below warning thresholds (at any site on Unguja). Significant differences are marked with 2 stars (**). Not marked are those for which no information about significance level was given by authors.

	Catch data Unguja	**Fishermen's**	**Ecological surveys**
Indicator (Threshold)	Catch (0.5 C_{max})	CPUE (n.a.)	Biomass/Abundance (B_{50}/N_{50})
General nearshore resources	Developed	-	
Siganidae	Fully exploited	-	$B < B_{50}$ [**2]
Scaridae	Developed	Overfished	$B < B_{50}$ [**1,2]
Lethrinidae	Developed	-	-
Serranidae	Developed	Overfished	-
Mullidae	Developed	-	$B < B_{50}$ [1]
Lutjanidae	Developed	Overfished	$B < B_{50}$ [1]
Mugilidae	Developed	-	-
Clupeioids	Fully exploited	-	-
Sardines	Fully exploited	Overfished	-
Mackerels	Fully exploited	-	-
Carangidae	Fully exploited	Overfished	-
Tuna-like fishes	Developed	-	-
Marlins, Sailfishes, etc.	Developed	-	-
Sphyraenidae	Developed	-	-
Sharks and Rays	Fully exploited	-	-
Molluscs (Octopus and Squids)	Developed	-	-
Shells (Gastropods and Bivalves)	-	Overfished	$N < N_{50}$ [6]
Sea cucumber	-	Overfished	$B < B_{50}$ [5]
Lobsters	Fully exploited (Recovering from being overexploited/depleted)	Overfished	-
Acanthuridae	-	-	$B < B_{50}$ [**1,2]
Balistidae	-	-	$B < B_{50}$ [1]
Chaetodontidae	-	-	$B < B_{50}$ [1]
Pomacanthidae	-	-	$B < B_{50}$ [1]

[1]*McClanahan et al., 1999;* [2]*Lokrantz et al., 2010;* [5]*Nordlund et al., 2011;* [6]*Fröcklin et al., 2014*

This review suggests that the highly vulnerable invertebrate groups lobsters and sea cucumbers together with several commercially important bivalve and gastropod species are in danger of being overexploited. In fact, the long period of overexploited/collapsed catches of lobsters from the nearshore area of Unguja and the strong decline in exports together with the absence of lobsters in the inventories conducted in Chwaka Bay by Fröcklin et al. (2014) indicate a general state of overexploitation for this group. The studies by Eriksson et al. (2010) and (2012) also revealed that commercially important sea cucumber species are subject to a very high fishing pressure both in shallow and deeper waters across the island. While no information is available about annual catches of sea cucumber for Zanzibar, in Tanzania mainland the fishing authorities placed an export moratorium in order to protect this heavily exploited resource (Mgaya and Mmbaga, 2007). Eriksson et al. (2010) clearly point out that the fishery on Zanzibar shows several indicators for an unsustainable exploitation of sea cucumbers. Taking into consideration their status on the mainland and their high vulnerability, these findings strongly suggests that commercially important sea cucumber species are likewise experiencing a state of overexploitation.

Despite the export of these highly valuable groups, invertebrate collection is an important source for food security and has shown to become more and more important for the income of fishing communities. Although, fishing households are strongly depending on invertebrates for income and protein supply, invertebrate collection is not monitored yet. While beach recorders are mostly collecting vessel-based catches, the catch of invertebrate collectors is not included in data collection, this particularly holds true for invertebrate catches that serve as home consumption. Nevertheless, the vulnerability of many invertebrate species together with the observed collapse of lobster catches and the depletion of several gastropod, bivalve and sea cucumber species in the north and east coast of Unguja Island, demonstrates the importance of quantifying gleaning effort and estimating annual invertebrate catches. While the Department of Marine Fisheries Resources on Zanzibar classifies the catches of finfish in 16 different groups, the catches of invertebrates are classified in molluscs (Octopus/Squids) and lobsters (Palinuridae) only. Information of other invertebrates is not collected or is pooled into the broad category molluscs. Children, women and men are all participating in the collection of invertebrates. Furthermore, harvest behaviour is driven by a large number of factors and can be highly variable. Thus, it will be an impossible task to monitor all gleaning activities accurately. However, annual household interviews in fishing communities can help in identifying overall effort and together with the collection of information about monthly catches of a subsample of collectors this should provide rough estimates of annual invertebrate catches. Thereby, catches need be classified at least into bivalves, gastropods, crabs, sea cucumbers, lobsters, octopus and squids.

Sea cucumbers are strongly harvested below length at first maturity. Although minimum size restrictions are in place, Eriksson et al. (2010) have demonstrated that these regulations are often not known, pointing to weak enforcement. Furthermore, Fröcklin et al. (2014) have pointed out that there is no formal management regime that regulates the harvest of invertebrate species specifically. This highlights the lack of recognition of the importance and the vulnerability of Zanzibar's invertebrate resources.

An important fishing area that shows indication of overfishing is Chwaka Bay (Jiddawi, 2012). Questionnaires with fishermen reveal a perception of reduced catch rates and the overall catch of the bay has been reported to have dropped from 370 t in 2003 to 310 t in 2007, despite an increase in fishing effort from 1469 fisher in 2003 to 1871 fisher in 2007. Contrastingly, Aller et al. (2014) found relatively high mean fish density in Chwaka Bay compared to the protected sites (i.e. Nungwi/Fumba). Nevertheless, due to the large standard deviation of the mean fish densities these differences were not significant. Thus, future investigation is required to assess which resources of Chwaka Bay are fished below sustainable limits and what management actions could allow for both profitable catches and resource protection.

The reefs on the west coast of Unguja Island have been extensively studied regarding biomass, community and habitat indicators. It seems that current levels of fishing at Changuu, Chapwani, Pange and Bawe is reducing fish biomass and species richness below warning thresholds. However, the reviewed studies did not properly quantify fishing effort at the studied sites and as such cannot provide effort reference levels for fisheries managers. Unless a complete closure of these fishing grounds is the target, changes in indicators need to be related to manageable input measures such as mesh size, total fishing effort and gear-specific fishing effort. The findings of the study from Tyler et al. (2011) challenge the suitability of enforcing the ban of illegal fishing methods such as dragnets in order to reduce pressure on target resources, because their findings indicate no difference in any ecological indicators despite species richness. However, species richness and diversity are indicators that have shown to often vary much more under natural perturbations than fishing (Rochet and Trenkel, 2003). As such it is questionable if the observed difference was caused by the removal of illegal fishing methods. In contrast, in southern Kenya a strong positive effect of gear management on the biomass/abundance and mean length of target fish has been observed (McClanahan and Hicks, 2011). Further studies should, therefore, investigate the effect of the removal of illegal fishing methods in Menai Bay on overall fishing effort, catch rates, the fish community and the habitat.

Future ecological surveys aimed at detecting impacts of different levels of fishing, should ideally quantify site-specific annual fishing effort. In the absence of time series data ecological surveys, such as the ones presented here, often use protected sites to generate baseline information. However the use of time series of indicators can help in ruling out other potential site-specific effects. McClanahan et al. (1999) and Lokrantz et al. (2010) provide baseline data for some of the reefs on the West Coast. Thus, we suggest that future studies should a) monitor the same set of indicators using the same methodological approach and b) quantify site-specific fishing effort. This could then be used to set fishing effort related thresholds. One of the central problems is that ecological surveys are often highly costs intensive and there is the lack of financial and institutional capacity to conduct periodic surveys around Zanzibar. In contrast, the collection of fisheries data at 31 landing sites around the two main islands Unguja and Pemba is already in place (Kathib and Jiddawi, 2010). Thus, we encourage future research studies to collaborate with the DMFR and make use of the existing data collection. Well-conducted fisheries assessments have the advantage that they are able to quantify the status of populations and can provide fisheries managers with clear and tangible reference points. Combining such studies with ecological surveys can then help to quantify fisheries impacts beyond the level of target species and to adjust reference points accordingly.

Table 2.3a. Overview of the findings of the ecological surveys and invertebrate inventories conducted throughout Zanzibar. Presented are all differences in indicators for which quantitative information was provided by the authors. Significant differences, if reported by the authors, were marked with two stars (**), while non-significant outcomes were marked with one star (*). Some authors did not provide information or test whether or not observed differences were significant, these were not marked at all (Part I: Unguja West).

	Site	Habitat	Target resource	Biomass/Abundance (B_{50}/N_{50})	Species richness/diversity (SR_{90}/SD_{90})	Mean length (L_{70})	Reference Site
Unguja West	Chapwani[1]	Reef	Acanthuridae	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	-	Chumbe (Fully protected)
	Chapwani[1]	Reef	Balistidae	$B < B_{50}^{**}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Chapwani[1]	Reef	Chaetodontidae	$B < B_{50}^{**}$	$SR > SR_{90}^{*}$	-	Chumbe (Fully protected)
	Chapwani[1]	Reef	Labridae	$B > B_{50}^{**}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Chapwani	Reef	Lutjanidae	$B < B_{50}^{**}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Chapwani[1]	Reef	Mullidae	$B < B_{50}^{**}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Chapwani[1]	Reef	Pomacanthidae	$B < B_{50}^{**}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Chapwani[1]	Reef	Pomacentridae	$B > B_{50}^{*}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Chapwani[1]	Reef	Scaridae	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	-	Chumbe (Fully protected)
	Chapwani[1]	Reef	Siganidae	$B < B_{50}^{*}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)

Table 2.3a. (continued)

	Site	Habitat	Target resource	Biomass/Abundance (B_{50}/N_{50})	Species richness/diversity (SR_{90}/ SD_{90})	Mean length (L_{70})	Reference Site
Unguja West	Changuu[1,2]	Reef	Acanthuridae	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	$L > L_{70}^{*}$	Chumbe (Fully protected)
	Changuu[1]	Reef	Balistidae	$B < B_{50}^{**}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Changuu[1]	Reef	Chaetodontidae	$B < B_{50}^{**}$	$SR > SR_{90}^{*}$	-	Chumbe (Fully protected)
	Changuu[1]	Reef	Labridae	$B > B_{50}^{*}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Changuu[1]	Reef	Lutjanidae	$B < B_{50}^{*}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Changuu[1]	Reef	Mullidae	$B < B_{50}^{**}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Changuu[1]	Reef	Pomacanthidae	$B < B_{50}^{**}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Changuu[1]	Reef	Pomacentridae	$B < B_{50}^{**}$	$SR < SR_{90}^{*}$	-	Chumbe (Fully protected)
	Changuu[1,2]	Reef	Scaridae	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	$L < L_{70}^{**}$	Chumbe (Fully protected)
	Changuu[1]	Reef	Siganidae	$B > B_{50}^{*}$	$SR < SR_{90}^{*}$	$L > L_{70}^{*}$	Chumbe (Fully protected)
	Changuu[3]	Seagrass	Total fish	$N < N_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Changuu[3]	Seagrass	Algal herbivore fish	$N < N_{50}^{*}$	NA	-	Nungwi and Fumba (Remote /protected area)

Table 2.3a. (continued)

	Site	Habitat	Target resource	Biomass/Abundance (B_{50}/N_{50})	Species richness/diversity (SR_{90}/ SD_{90})	Mean length (L_{70})	Reference Site
Unguja West	Changuu[3]	Seagrass	Seagrass herbivore fish	$N < N_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Changuu[3]	Seagrass	Invertebrate feeder fish	$N < N_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Changuu[3]	Seagrass	Invertebrate/fish feeder fish	$N > N_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Changuu[3]	Seagrass	Omnivore fish	$N < N_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Changuu[3]	Seagrass	Herbivore fish	$N < N_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Bawe[2]	Reef	Excavators	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	$L < L_{70}^{**}$	Chumbe (Fully protected)
	Bawe[2]	Reef	Scrapers	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	$L > L_{70}^{*}$	Chumbe (Fully protected)
	Bawe[2]	Reef	Grazers	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	$L < L_{70}^{*}$	Chumbe (Fully protected)

Table 2.3a. (continued)

	Site	Habitat	Target resource	Biomass/Abundance (B_{50}/N_{50})	Species richness/diversity (SR_{90}/ SD_{90})	Mean length (L_{70})	Reference Site
Unguja West	Pange[2]	Reef	Excavators	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	$L < L_{70}^{**}$	Chumbe (Fully protected)
	Pange[2]	Reef	Scrapers	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	$L > L_{70}^{*}$	Chumbe (Fully protected)
	Pange[2]	Reef	Grazers	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	$L > L_{70}^{*}$	Chumbe (Fully protected)
	Nyange[2]	Reef	Excavators	$B > B_{50}^{**}$	$SR > SR_{90}^{**}$	$L > L_{70}^{**}$	Chumbe (Fully protected)
	Nyange[2]	Reef	Scrapers	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	$L > L_{70}^{*}$	Chumbe (Fully protected)
	Nyange[2]	Reef	Grazers	$B < B_{50}^{**}$	$SR < SR_{90}^{**}$	$L > L_{70}^{*}$	Chumbe (Fully protected)
	Mbweni[3]	Seagrass	Total fish	$N < N_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Mbweni[3]	Seagrass	Algal herbivore fish	$N < N_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Mbweni[3]	Seagrass	Seagrass herbivore fish	$N < N_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Mbweni[3]	Seagrass	Invertebrate feeder fish	$N < N_{50}^{**}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Mbweni[3]	Seagrass	Invertebrate/fish feeder fish	$N < N_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)

[1]McClanahan et al., 1999; [2]Lokrantz et al., 2010; [3]Aller et al., 2014

Table 2.3b. Overview of the findings of the ecological surveys and invertebrate inventories conducted throughout Zanzibar. Presented are all differences in indicators for which quantitative information was provided by the authors. Significant differences, if reported by the authors, were marked with two stars (**), while non-significant outcomes were marked with one star (*). Some authors did not provide information or test whether or not observed differences were significant, these were not marked at all (Part II: Unguja South, North, East and Pemba).

	Site	Habitat	Target resource	Biomass/Abundance (B_{50}/N_{50})	Species richness/diversity (SR_{90}/ SD_{90})	Mean length (L_{70})	Reference Site
Unguja South	Fumba[4]	Intertidal zone	Sea cucumber	-	SD < SD_{90}	-	Chumbe (Fully protected)
Unguja North	Nungwi[5]	Seagrass	Total invertebrates	N < N_{50} **	SR < SR_{90} **	-	Nungwi (Remote site)
	Nungwi[5]	Seagrass	Bivlaves	B > B_{50}	SR < SR_{90}	-	Nungwi (Remote site)
	Nungwi[5]	Seagrass	Gastropods	B > B_{50}	SR < SR_{90}	-	Nungwi (Remote site)
	Nungwi[5]	Seagrass	Crustacea	B < B_{50}	SR < SR_{90}	-	Nungwi (Remote site)
	Nungwi[5]	Seagrass	Sea cucumber	B < B_{50}	SR > SR_{90}	-	Nungwi (Remote site)

Table 2.3b. (continued)

	Site	Habitat	Target resource	Biomass/Abundance (B_{50}/N_{50})	Species richness/diversity (SR_{90}/ SD_{90})	Mean length (L_{70})	Reference Site
Unguja East	Chwaka[3]	Seagrass	Total fish	$B > B_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Chwaka[3]	Seagrass	Detritivorous fish	$B > B_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Chwaka[3]	Seagrass	Herbivorous fish	$B > B_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Chwaka[3]	Seagrass	Invertivorous fish	$B > B_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Chwaka[3]	Seagrass	Nekt-Invertivorous fish	$B > B_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Chwaka[3]	Seagrass	Nektonic fish	$B > B_{50}^{*}$	NA	-	Nungwi and Fumba (Remote/protected area)
	Chwaka[6] (2010)	Seagrass	Total invertebrates	$N > N_{50}^{**}$	-	-	Chwaka (2005)
	Chwaka[6] (2010)	Seagrass	Gastropods	$N < N_{50}^{**}$	-	-	Chwaka (2005)
	Chwaka[6] (2010)	Seagrass	Bivalves	$N < N_{50}^{**}$	-	-	Chwaka (2005)

Table 2.3b. (continued)

	Site	Habitat	Target resource	Biomass/Abundance (B_{50}/N_{50})	Species richness/diversity (SR_{90}/ SD_{90})	Mean length (L_{70})	Reference Site
Pemba	Misali[7]	Reef	Total fish	$N > N_{50}^{*}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Lethrinidae	$N > N_{50}^{**}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Lutjanidae	$N > N_{50}^{**}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Mullidae	$N > N_{50}^{**}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Holocentridae	$N < N_{50}^{**}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Scaridae	$N < N_{50}^{**}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Acanthuridae	$N > N_{50}^{*}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Ballistidae	$N > N_{50}^{*}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Caesionidae	$N < N_{50}^{*}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Carangidae	$N < N_{50}^{*}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Siganidae	$N > N_{50}^{*}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Haemulidae	$N > N_{50}^{*}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Labridae	$N > N_{50}^{*}$	-	$L > L_{70}^{*}$	Misali (Fully protected)
	Misali[7]	Reef	Serranidae	$N < N_{50}^{*}$	-	$L > L_{70}^{*}$	Misali (Fully protected)

[3]Aller et al., 2014; [4]Eriksson et al., 2010; [5]Nordlund et al., 2011; [6]Fröcklin et al., 2014; [7]Daniels et al., 2003

Table 2.3c. Overview of the findings of the ecological surveys and invertebrate inventories conducted throughout Zanzibar. Presented are all differences in indicators for which quantitative information was provided by the authors. Significant differences, if reported by the authors, were marked with two stars (**), while non-significant outcomes were marked with one star (*). Some authors did not provide information or test whether or not observed differences were significant, these were not marked at all (Part II: Different locations).

	Site	Habitat	Target resource	Biomass/Abundance (B_{50}/N_{50})	Species richness (SR_{90})	Mean length (L_{70})	Reference Site
Different Locations	Pange/Bawe[8]	Reef	Total fish	-	$SR < SR_{90}^{**}$	-	Chumbe (Fully protected)
	Pange/ Changuu/ Bawe/Chawacha/Paje[9]	Reef	Total fish	$B > B_{50}^{*}$	$SR < SR_{90}^{**}$	$L > L_{70}^{*}$	Menai Bay (Partially protected)
	Pange/ Changuu/ Bawe/Chawacha/Paje[9]	Reef	Herbivorous fish	NA	$SR < SR_{90}^{**}$	NA	Menai Bay (Partially protected)
	Pange/ Changuu/ Bawe/Chawacha/Paje[9]	Reef	Invertivorous fish	NA	$SR < SR_{90}^{**}$	NA	Menai Bay (Partially protected)
	Pange/ Changuu/ Bawe/Chawacha/Paje[9]	Reef	Nekt-Invertivorous fish	NA	$SR < SR_{90}^{**}$	NA	Menai Bay (Partially protected)
	Pange/ Changuu/ Bawe/Chawacha/Paje[9]	Reef	Nektonic fish	NA	$SR < SR_{90}^{**}$	NA	Menai Bay (Partially protected)
	Pange/ Changuu/ Bawe/Chawacha/Paje[9]	Reef	Detritivorous fish	NA	$SR < SR_{90}^{**}$	NA	Menai Bay (Partially protected)

[8]Tyler et al., 2009; [9]Tyler et al., 2011

CHAPTER III - Fisheries Assessment of Chwaka Bay

Chapter III.

Fisheries assessment of Chwaka Bay (Zanzibar) – following a holistic approach

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Abstract

The typical East African multispecies fishery of Chwaka Bay (Zanzibar) was investigated by assessing the current level of exploitation and stock size of six key target fish stocks in order to investigate long-term concerns of overexploitation of the Bay's resources. An assessment of the length frequencies of the catches over an annual cycle revealed that (1) the exploitation rate of three out of six target species exceeds recommended levels ($E_{0.1}$); (2) despite high juvenile retention rates, fishing mortalities are highest for specimens above length at first maturity. Two management measures are discussed: (1) increase in mesh size and (2) closure of the destructive dragnet fishery. The first option only seems feasible, if the radius of the fishery was increased to capture larger specimens outside the shallow bay area. The second would leave approximately 450 fishermen unemployed. The redistribution of dragnet fishermen to other gears would lead to a substantial increase in the number of boats, which would create spatial use and pollution problems. The general state of the bay`s fishery is considered to represent a "full exploitation to slight overexploitation scenario" with no scope for expansion.

Keywords: stock assessment, bay resources, fisheries management, artisanal fishery, multispecies fishery

3.1. Introduction

In large parts of the Western Indian Ocean artisanal fisheries are the major provider of income and protein supply for local communities (Walmsley et al., 2006). Characteristics such as open access and multigear use together with a lack of effective enforcement in these areas put great pressure on fisheries resources (Najmudeen and Sathiadhas, 2008; Walmsley et al., 2006). Since communities are directly depending on the harvest of these resources, a decrease in catch often leads to an increase in effort or to the use of more efficient and eventually destructive gears (Jiddawi and Ohman, 2002; McClanahan and Hicks, 2008). Consequently, such fisheries are often considered unsustainable leading to Malthusian overexploitation (Pauly, 1994). However, a decrease in catch per unit of effort and a shift towards new gears and smaller mesh sizes not necessarily reflects a situation of overfishing as Kolding and Van Zwieten (2014) have pointed out. The authors demonstrated, inter alia for the swamp and lake fishery of Bangweulu and Meru, that despite shifts towards the usage of multiple gears and smaller mesh sizes under a disobedience of regulations, total fisheries catches remained stable and the annually obtained yield was close to the potential long term maximum sustainable yield for the studied ecosystem.

The fishery of Chwaka Bay, located at the east coast of Zanzibar (Tanzania) in the Western Indian Ocean, is a typical example for an artisanal, multigear fishery. The local community is highly dependent on fish as a source of income and protein supply (Jiddawi and Lindström, 2012). Consequently, along with an increase in tourism and population density on Zanzibar (de la Torre-Castro and Lindström, 2010; Lange and Jiddawi, 2009), the Chwaka Bay fishery has seen a steady increase in effort during the last years (Kathib and Jiddawi, 2010) and fishermen report that resources are decreasing. This decrease is said to be associated with high fishing effort, destructive fishing gears and the damage of coral and seagrass habitats in the bay (de la Torre-Castro, 2012; Tobey and Torell, 2006). The use of small mesh size has been identified as highly unsustainable as large parts of the catch are comprised of juveniles (de la Torre-Castro and Rönnbäck, 2004) leading to the presence of growth and recruitment overfishing (de la Torre-Castro et al., 2014). The use of dragnets in the bay is destroying corals and seagrasses by pulling the net over the sea floor (Jiddawi and Ohman, 2002). It has been shown that, although these nets are banned by the Chwaka Bay by-law (2001), their use is steadily increasing (de la Torre-Castro, 2012). This situation is believed to be highly unsustainable causing overexploitation of Chwaka Bay's resources (de la Torre-Castro and Rönnbäck, 2004; Nordlund et al., 2011). However, research focusing on the fishery of Chwaka Bay is dominated by descriptive studies (de la Torre-Castro and Lindström, 2010; Eklöf et al., 2012), analysis of management and governance aspects (de la Torre-Castro and Lindström, 2010; de la Torre-Castro, 2012; Gustavsson et al., 2014; Tobey and Torell, 2006), and studies focusing on the importance of Chwaka Bay's seagrass meadows for fisheries catches and revenues (de la Torre-Castro and Rönnbäck, 2004). Despite this research effort only few studies provide information about catch trends and the status of individual resources. Jiddawi and Ohman (2002), for instance, showed a decline in the catch of reef fish, lobsters as well as small and large pelagic fish between 1994 and 1998. However, no up to date study is available on trends in landings for these fish groups. Furthermore, they represent short-term isolated catch trends, which are not set in relation to abundance levels, maximum sustainable yield or any other biological reference point. Declines in resource abundance have been suggested recently for the sea cucumber and invertebrate fishery (Eriksson et al., 2010; Fröcklin et al., 2014). Invertebrates are sessile and have a low motility, which makes them prone to overexploitation (Perry et al., 1999) and thus cannot be used as proxies for the status of Chwaka Bay's nearshore resources. Observations made by local fishermen on the decline in individual catches directly relate to a decrease in catch per unit of effort, which has been shown to be insufficient in assessing the status of fisheries resources (Maunder et al., 2006).

It thus seems that Chwaka Bay's fisheries are in need of a proper biological assessment, in order to derive sound management measures for a long-term sustainable use of the bay's resources. The objectives of this paper are to assess the status of Chwaka Bay's fisheries resources to 1) evaluate the assumption of overexploitation and 2) to discuss management options for improving their sustainable use. The study is based on the analysis of the catch length frequency composition data of the dominant target species in order to 1) assess the size spectrum of the catch 2) estimate juvenile retention rates 3) assess fishing mortality and exploitation status, 4) compare current exploitation rates with biological reference points and 5) estimate stock size by using cohort analysis.

3.2. Material and Methods

3.2.1. Study site and data collection

The study site Chwaka Bay is located at the east coast of Unguja Island, Zanzibar (6°02'6°13'S, 39°24'39°36'E, Fig. 1.1). The bay consists of dense seagrass meadows, large mangrove stands along the shore and a fringing reef, protecting the bay proper. The sea surface temperature ranges from 25 - 31° C (Jiddawi and Lindström, 2012) and the depth ranges from 3 m in the south of the bay to 20 m around the outer reefs. The bay is one of the main fishing grounds of the island with the majority of the fishery concentrating in the bay and around the reefs, covering an area of approximately 124 km^2. The fishing gears operating in the area are basket traps, dragnets, handlines, spears and, to a minor extend, longlines and gillnets. Data collection took place on 18 days per month from January – June and September – December 2014 at the three mayor landing sites: Chwaka village, Marumbi village and Uroa village (Fig. 3.1.). Specimens were identified to species level using identification keys for the region (Smith and Heemestra, 1986; Anam and Mostarda, 2012). Specimens for size measurements were obtained from fisheries catches, measured to the nearest 0.5 cm total length (TL) and later grouped into 1 cm classes. The following six species dominated the catches: *Siganus sutor* (Valenciennes 1835), *Lethrinus borbonicus* (Valenciennes 1830), *Leptoscarus vaigiensis* (Quoy and Gaimaird 1824), *Lethrinus lentjan* (Lacepède 1802), *Scarus ghobban* (Forsskål 1775) and *Lutjanus fulviflamma* (Forsskål 1775). They represented approximately 59 % of the total abundance (in numbers) of the catch. Consequently, these species were chosen for further stock assessment.

3.2.2. Size spectrum and juvenile retention rates

A size spectrum of 13 out of 44 identified osteichthyes families was analysed. These 13 families have been chosen as they contribute 89 % to the abundance (in numbers) in the overall catch. Total length of the individuals was first grouped into 5 cm classes and then plotted against the natural logarithm of its abundance for visualization of the size frequency distribution of the fish caught by the fishery. Additionally, juvenile retention rates were calculated as the proportion of fish in the catches that were smaller than length at first maturity (L_{mat}). Values for L_{mat} were obtained from Mangi and Roberts (2006) and Venkataramani and Jayakumar (2006) and values given in standard length were converted to total length using length-length relationships from FishBase (Froese and Pauly, 2015).

3.2.3. Growth parameters

Monthly length frequency distributions were constructed to estimate growth parameters using the *ELEFAN I* routine as implemented in the program package *FiSAT II* (Gayanilo et al., 1994). This routine is a nonparametric method and identifies the growth function that best fits the restructured length-frequencies using the Bertalanffy equation. The routine assigns to each growth curve a value for the goodness of fit (*Rn*, ranging from 0 to 1) by dividing the total number of peaks and troughs, the curve runs through, by the overall number of peaks and troughs available. The parameter range for L_∞ used to find the best growth curve was set from the maximum size found in the sample (L_{max}) to $L_{max}/0.9$, while for *K* a wide range from 0.1 to 2 was used. In addition the growth performance index *phi* was calculated (Pauly and Munro, 1984) and an arbitrary value of zero for t_0 was used for estimating *K* and L_∞.

3.2.4. Mortality and exploitation rates

The length converted catch curve as implemented in *FiSAT II* was used to estimate the total instantaneous mortality rate. A regression was fitted to the points of the catch curve, where L' is the first point of the curve used for the regression analysis and represents the smallest length at which probability of capture is 1. The slope of the regression line represents the estimate of the total mortality rate *Z*. Natural mortality *M* was then calculated with Pauly's formula (1980) using a mean habitat temperature of 28° C. The fishing mortality and exploitation rate were calculated as:

$$F = Z - M \qquad (1)$$

$$E = \frac{F}{Z} \qquad (2)$$

3.2.5. Yield per recruit analysis and effort estimation

The relative yield and biomass-per-recruit model (*Y/R* and *B/R*) of Beverton and Holt (1964) as implemented in *FiSAT II* was used to estimate biological reference points and optimal length at first capture (L_{opt}). Different combinations of *E* and length at first capture (L_c) are simulated to estimate the expected lifetime yield of a cohort. The input parameter *M/K* and L_∞ were derived from the *ELEFAN* I analyses, assuming that natural mortality is constant throughout the harvest lifetime of the cohort. Different L_c-values were used to estimate L_{opt} that generated the highest yield possible.

Values for exploitation rates were calculated, which a) generate the highest maximum yield (E_{max}), b) reduce the unexploited biomass by 50 % only ($E_{0.5}$) and c) result in a relative yield per recruit of one-tenth of the angle of the yield curve at its origin ($E_{0.1}$).

In order to assess the magnitude of effort reduction that has to be implemented to reach the target exploitation of $E_{0.1}$, $F_{0.1}$ was calculated by

$$F_{0.1} = \frac{E_{0.1} * M}{1 - E_{0.1}} \qquad (3)$$

and the catchability coefficient q was calculated using the current fishing mortality and the current effort (boat h^{-1}), which was then used to calculate the fishing effort under a fishing mortality of $F_{0.1}$:

$$F = qf \qquad (4)$$

3.2.6. Cohort analysis

Jones cohort analysis (1984) was also conducted to reconstruct the standing biomass of the stock and to estimate fishing mortality for different length groups. Input values for L_∞ and K were taken from the *ELEFAN I* analysis. The annual mean value of F derived through the length converted catch curve was used as an estimate for the fishing mortality of the last length group. The last lengths groups, which were represented in the catches only in low numbers, were grouped into plus groups. Biomass of the different length groups was then calculated with the length-weight relationship formula. Values for the constant (a) and the exponent (b) were taken from FishBase (Froese and Pauly, 2015).

3.3. Results

3.3.1. Size spectrum and juvenile retention rates

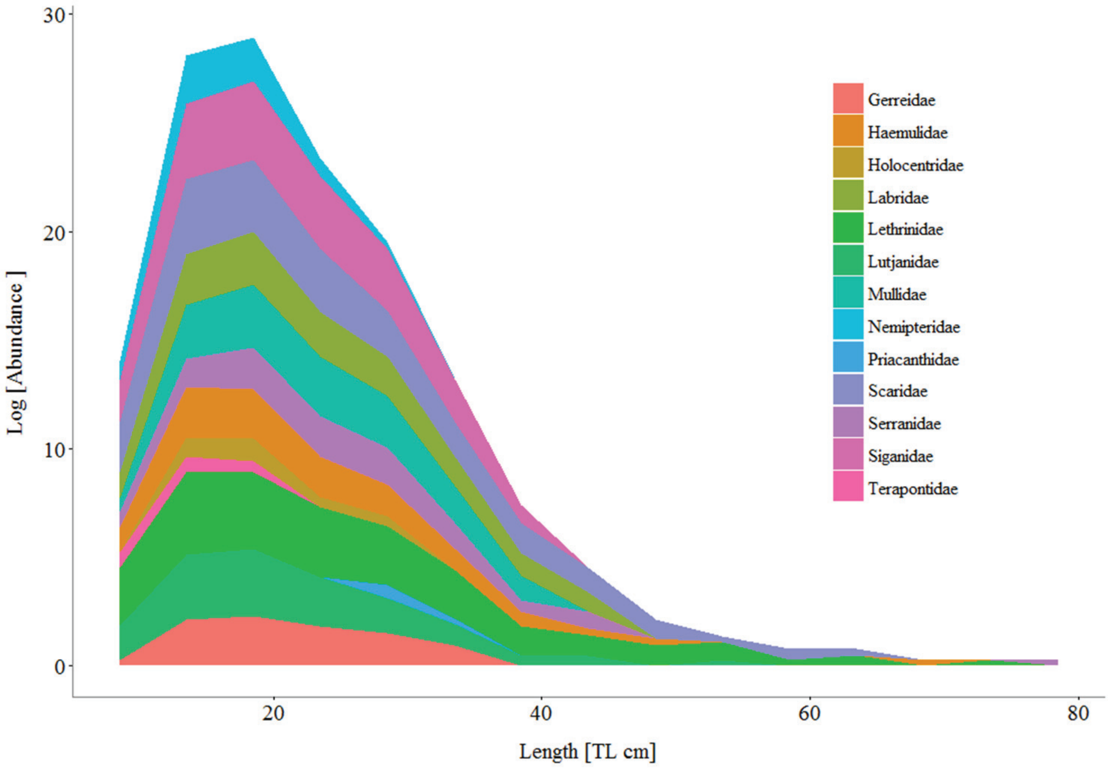

Fig. 3.1. Size spectrum of 13 target families found in the catches of Chwaka Bay.

The size spectrum of 13 target fish families found in the catch is depicted in Fig. 3.2. Size distribution ranges from 7.5 to 78.5 cm with a peak at 11 to 21 cm. Table 3.1. presents the calculated proportion of juveniles in the catches of Chwaka bay based on length at first maturity estimates. Juvenile retention rates are exceeding 80 % for all species, except *L. vaigiensis*, which showed a considerably lower value (41.9 %).

Table 3.1. Juvenile retention rates in the catches of the six key species found in 2014, based on literature values for length at first maturity ([1]Mangi and Roberts 2006; [2]Venkataramani and Jayakumar 2006).

Species	Length at first maturity [TL cm]	Juvenile retention rate [%]
Siganus sutor	20.2[1]	82
Lethrinus borbonicus	14.8[1]	86.8
Leptoscarus vaigiensis	15.1[1]	41.9
Lethrinus lentjan	20.3[1]	89
Lutjanus fulviflamma	15.6[1]	98.7
Scarus ghobban	40[2]	99

3.3.2. Growth parameters

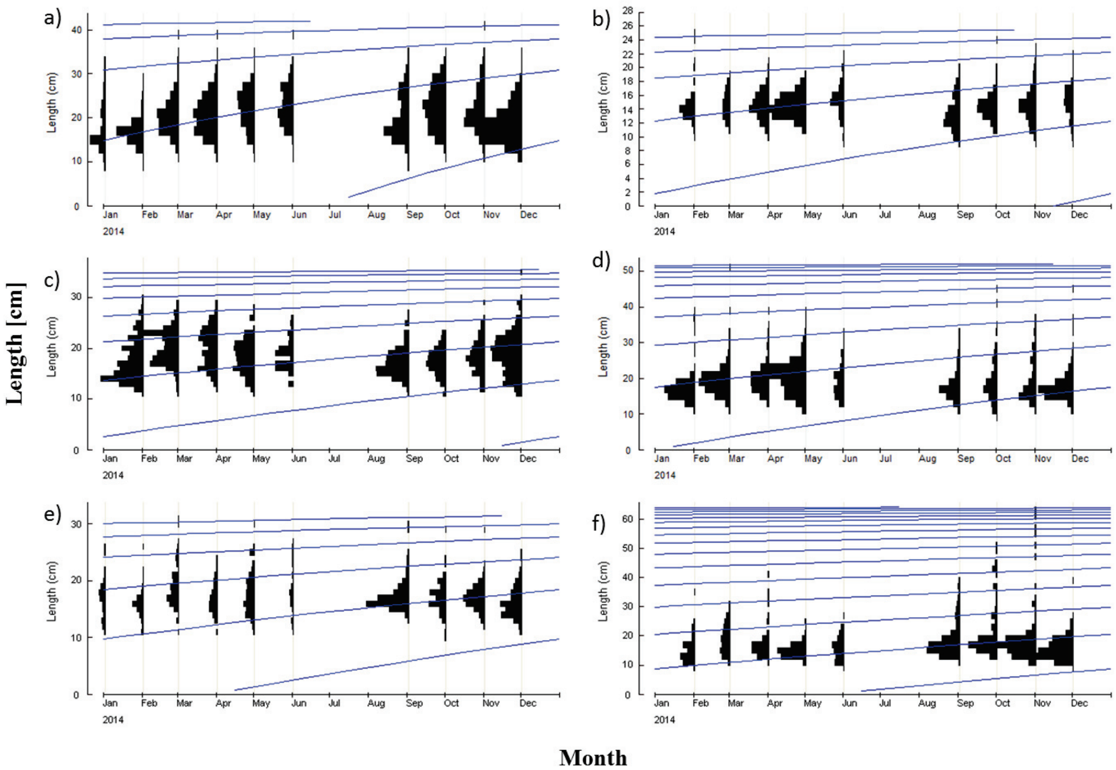

Fig. 3.2. Length frequency distributions of catches (TL cm) with the corresponding growth curves estimated by ELEFAN I a) S. sutor, b) L. borbonicus, c) L. vaigiensis, d) L. lentjan, e) L. fulviflamma, f) S. ghobban.

For each of the six species a monthly number of 146 - 1370 individuals were measured at the landing sites. The Bertalanffy growth curves computed and imposed on the length frequency histograms are shown in Fig. 3.3. The corresponding best fit of growth parameters with their *Rn*-values, and the growth performance indices are shown in Table 3.2. For all species the final L_∞-values were not higher than $L_{max}/0.9$.

Table 3.2. Growth parameters derived by ELEFAN I (L_∞, K) with the corresponding value of fit (Rn), mortality and exploitation rates estimated from the length converted catch curve (Z, M, F, E) with the coefficient of determination (R^2).

Species	**L_∞ [Tl cm]**	**K [yr^{-1}]**	**Φ**	**Rn**	**Z [yr^{-1}]**	**M [yr^{-1}]**	**F [yr^{-1}]**	**E**	**R^2**
Siganus sutor	43.8	0.8	3.19	0.325	3.73	1.40	2.33	0.62	0.95
Lethrinus borbonicus	27.6	0.52	2.6	0.237	3.56	1.20	2.36	0.66	0.99
Leptoscarus vaigiensis	37	0.39	2.73	0.251	2.09	0.92	1.18	0.56	0.95
Lethrinus lentjan	53	0.4	3.1	0.314	2.64	0.84	1.80	0.68	0.97
Lutjanus fulviflamma	34.3	0.44	2.71	0.256	2.12	1.01	1.10	0.52	0.87
Scarus ghobban	64.4	0.24	3	0.368	1.07	0.57	0.49	0.46	0.83

Table 3.3. Biological reference points ($F_{0.1}$, $E_{0.1}$, $E_{0.5}$ and E_{max}), current and optimal length at first capture (L_c and L_{copt}) estimated from yield per recruit analysis. Mean annual biomass, annual yield and mean fishing mortality (F_{mean}) derived from cohort analysis.

Species	**L_c[Tl cm]**	**L_{opt}[Tl cm]**	**$E_{0.1}$**	**$E_{0.5}$**	**E_{max}**	**$F_{0.1}$**	**Mean Biomass [t yr^{-1}]**	**Yield [t yr^{-1}]**	**F_{mean}**
Siganus sutor	15.6	30.1	0.47	0.31	0.57	1.25	65.01	127.34	2.05
Lethrinus borbonicus	13.4	21.1	0.56	0.35	0.7	1.51	11.14	12.51	1.38
Leptoscarus vaigiensis	17.3	17.3	0.61	0.35	0.75	1.40	21.61	16.77	1.01
Lethrinus lentjan	12.6	42.8	0.4	0.28	0.48	0.57	17.47	23.24	1.07
Lutjanus fulviflamma	13.8	13.1	0.51	0.33	0.65	1.04	15.87	13.98	0.85

3.3.3. Mortality and exploitation rates

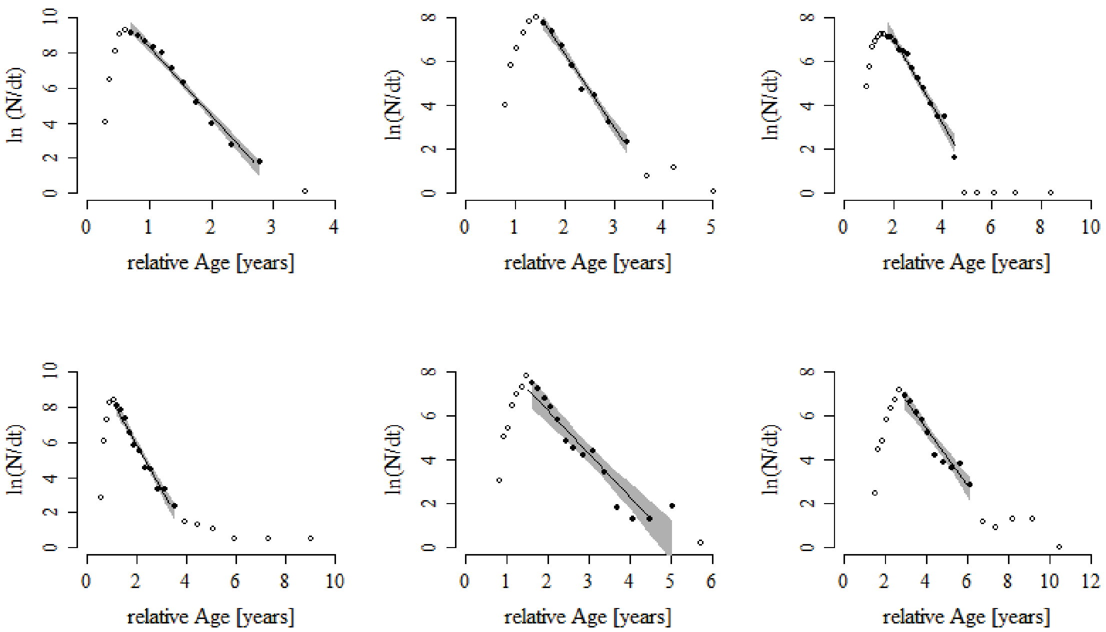

Fig. 3.3. Length converted, linearized catch curve with respective confidence intervals (0.95) of a) S. sutor, b) L. borbonicus, c) L. vaigiensis, d) L. lentjan, e) L. fulviflamma and S. ghobban. The black dots depict the points used in the regression analysis.

The length-converted catch curve and the corresponding linear regression computed to estimate total mortality for all six target species is shown in Fig. 3.4. The points that greatly deviate from the straight regression line and exhibit very low (ln) abundance values, were not included in the regression analysis, because these older/longer specimens are considered not efficiently selected by the multigear fishery inside the bay. Moreover, the uncertainty in the relationship between length and age, increases when approaching L_∞ (Gayanilo and Pauly, 1997). Resulting mortality and exploitation rates and the coefficient of determination of the regression analysis are listed in Table 3.2. Total mortality rates varied between species ranging from 1.07 to 3.73 yr^{-1}. Fishing mortalities for the two most abundant species *S. sutor* and *L. borbonicus* were twice as high as those of *L. vaigiensis* and *L. fulviflamma* and fourfold the fishing mortality of *S. ghobban*. The highest exploitation rates were found for the two emperor species *L. lentjan* and *L. borbonicus*, followed by the most abundant species in the catches *S. sutor*. *L. vaigiensis* and *L. fulviflamma* showed lower *E*-values and *S. ghobban* had the lowest exploitation rates.

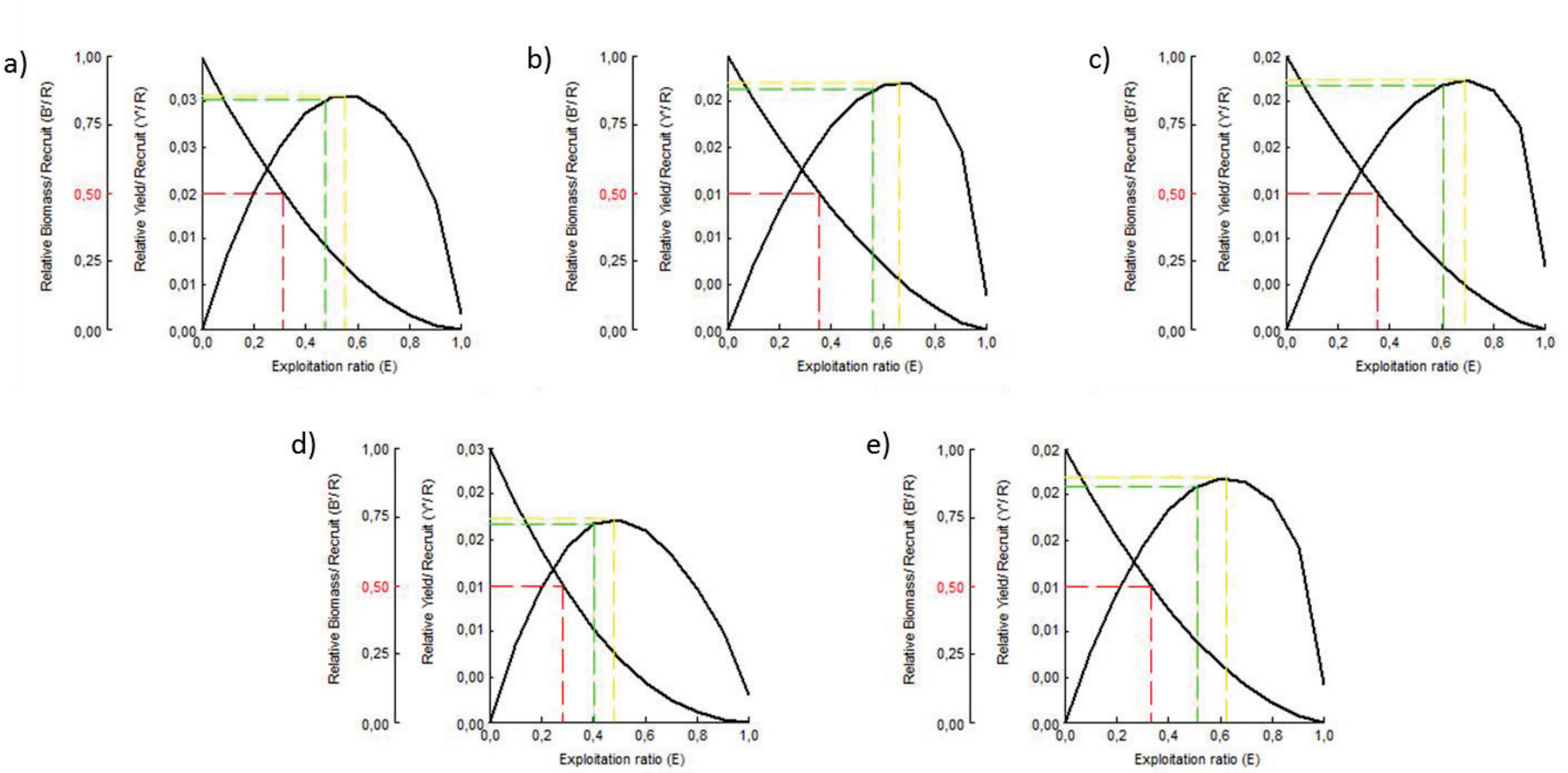

Fig. 3.4. Relative yield per recruit and biomass per recruit using selection-ogive as implemented in FiSAT II of a) S. sutor, b) L. borbonicus, c) L. vaigiensis, d) L. lentjan and e) L. fulviflamma. The dashed-lines represent the biological reference points: the maximum exploitation ratio (yellow lines, E_{max}), exploitation ratio that reduces the unexploited biomass by 50 % (red lines, E_{50}) and the exploitation ratio that results in a relative yield per recruit of one-tenth of the angle of the yield curve at its origin (green lines, $E_{0.1}$).

3.3.4. Yield per recruit and biomass per recruit analysis

The catch curve of *S. ghobban* was not suitable for backward extrapolation to estimate probabilities of capture. Consequently, *Y/R* and *B/R* analyses were only conducted for *S. sutor*, *L. borbonicus*, *L. vaigiensis*, *L. lentjan* and *L. fulviflamma*. *Y/R* and *B/R* as a function of the exploitation of the stock are depicted in Fig. 3.5. Analyses suggest that current exploitation rates substantially exceed the recommended $E_{0.1}$ level, except for *L. vaigiensis* and *L. fulviflamma* (Table 3.3.). For *S. sutor* and *L. lentjan* exploitation levels are beyond the maximum exploitation ratio, E_{max}.

Results show that yield per recruit is maximized (using $E_{0.1}$ as maximum) for *L. vaigiensis* at current L_c and current exploitation rates. To maximize yield per recruit for *L. fulviflamma*, *L. lentjan* and *S. sutor* length at first capture has to be increased to 15.9 cm, 24.3 cm and 21.3 cm and exploitation rates have to be decreased to 0.56, 0.56 and 0.55, respectively.

If the current fishing mortality pattern is maintained, length at first capture has to be increased for *S. sutor*, *L. borbonicus* and *L. lentjan* to 30.1 cm, 21.1 cm and 42.8 cm respectively (Table 3.3.), as exploitation rates should not exceed $E_{0.1}$.

The current daily number of boats operating in the bay (including all gears) was estimated to be 164 with an average fishing time of 3.2 h (463.2 boats h^{-1} day^{-1}). Given the current L_c, estimations of target fishing mortality $F_{0.1}$ (Table 3.3.) resulted in a decrease of effort to 77 boats for *S. sutor*, 92 boats for *L. borbonicus* and 45 boats for *L. lentjan*.

3.3.5. Cohort analysis

The results of the Jones cohort analysis are shown in Fig. 3.6. *S. ghobban* was excluded from the analysis, as natural mortality exceeded fishing mortality, which implies that the calculation errors could be substantial (Gayanilo and Pauly, 1997). Fishing mortalities vary between sizes and show peaks between 26 cm and 36 cm for *S. sutor*, 15.5 cm and 17.5 cm for *L. borbonicus*, 21 cm and 23 cm for *L. lentjan*, 15.5 cm and 20.5 cm for *L. fulviflamma* and at 28.5 cm for *L. vaigiensis*. The part of the mortality attributed to fishing ranges from 34 % to 51 % for the different species. Estimated annual mean stock biomass in the fishing area differed highly (Table 3.3.) between species with highest values for *S. sutor*, intermediate values for *L. vaigiensis*, *L. lentjan* and *L. fulviflamma* and lowest values for *L. borbonicus*.

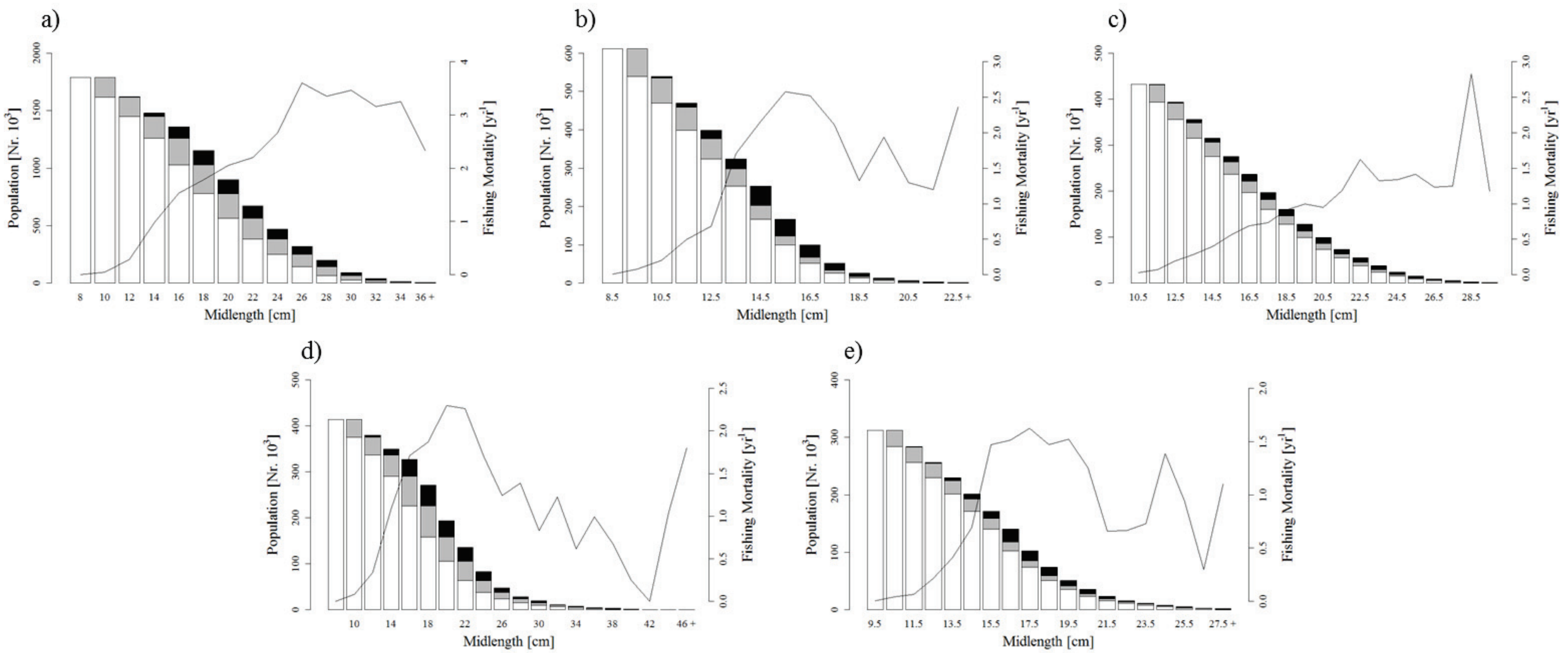

Fig. 3.5. Results of the Jones cohort analysis. Fishing mortalities (black line) and population numbers (stacked bars) for the survivors (white), the catch (black) and the losses due to natural mortality (grey) are plotted over the length groups for a) S. sutor, b) L. borbonicus, c) L. vaigiensis, d) L. lentjan and e) L. fulviflamma.

3.4. Discussion

3.4.1. Growth and mortality

The *phi*-values derived from the estimated growth parameters of *S. sutor* (*phi* 3.19), *L. fulviflamma* (*phi* 2.71), *L. borbonicus* (phi 2.6) and *S. ghobban* (*phi* 3) were similar to those reported in previous studies by Ntiba and Jaccarini (1988) conducted in Kenyan marine inshore waters (*S. sutor*, *phi* 3.06), Grandcourt et al. (2006) conducted in the southern Arabian Gulf (*L. fulviflamma, phi* 2.75), ElGanainy and Amin (2012) conducted on populations of the Gulf of Suez (*L. borbonicus*, *phi* 2.67) and Taylor and Choat (2014) conducted in the waters of Micronesia (*S. ghobban*, *phi* 3.13). For *L. lentjan* the *phi*-value 3.1 differed slightly to the value obtained by Grandcourt et al. (2011, *phi* 2.91), who assessed the growth parameters of *L. lentjan* in the Southern Arabian Gulf. However, the *K*-value found in their study (0.7) might overestimate growth, as maximum reported age of their samples obtained by otolith readings was 11 years. Also L_{inf}-values found by Grandcourt et al. (2011) are comparatively low (33.9 cm fork length), since maximum reported length of *L. lentjan* is around 52 cm (Froese and Pauly, 2015).

3.4.2. Size spectrum of fisheries catches and current exploitation pattern

The isolated size spectrum of Chwaka Bay's fisheries catches, has led to the assumption that the current fishing pattern leads to growth overfishing (de la Torre-Castro and Rönnbäck, 2004; de la Torre-Castro et al., 2014). The size spectrum of the catch shows a strong concentration of Chwaka Bay's fishing effort on the small, abundant part of the fish community. The estimated juvenile retention rates for 5 out of 6 species is above 80 %. Similarly, length at first capture under the current fishing pattern is 1.5 to 3 fold below the optimum for *S. sutor*, *L. borbonicus* and *L. lentjan*. A similar situation is found in the Southern Kenyan fishery, where *S. sutor* and *L. lentjan* are assumed to have reached a state of overexploitation, based on high juvenile retention rates (>80 %) in combination with exploitation rates over 0.5 (Hicks and McClanahan, 2012; Kaunda-arara et al., 2003). However, in the case of Chwaka Bay, we see that despite the high juvenile retention rates and L_c-values being below L_{opt}-values for *S. sutor* and *L. borbonicus* fishing mortality peaks at sizes above L_{mat}. For sizes below L_{mat} exploitation rates do not exceed reference points. Only *L. lentjan* experiences high exploitation values at sizes below L_{mat}. This shows that high proportions of immature fish in the catches, not necessarily indicates growth overfishing.

Yield per recruit analysis revealed that the current exploitation rates for *S. sutor* and the emperor *L. lentjan,* exceed the biological reference point E_{max}, suggesting an overexploitation of these stocks. The exploitation status for *L. lentjan* on the Southern coast of Kenya is likewise indicating an overfished state of this species (0.82, Hicks and McClanahan, 2012). In contrast, exploitation estimates for *S. sutor* in Kenyan coastal waters seem to be more moderate (0.53, Hicks and McClanahan, 2012; 0.56, Kaunda-arara et al., 2003) and considering reference points found in this study, these stocks do not seem to have reached an overexploitation state. However, the lack of appropriate reference points led the authors to the conclusion that estimates indicate an overharvesting of *S. sutor*. The data poor situation in East Africa (van der Elst et al., 2005; Walmsley et al., 2006), makes it difficult to appropriately assess fisheries stocks, however biological reference points are necessary, when evaluating their status.

S. sutor is by far the most important species for the fishery of Chwaka Bay. Its annual yield is twice as high as the yield of the other 5 species combined. Hence fishing pressure on *S. sutor* should not be increased since a collapse of the stock would lead to significant losses in economic rent of the fishermen (up to 20 %). For *L. borbonicus* and *L. vaigiensis* current exploitation rates are lower than E_{max}. However, calculated reference points are quite high and should be considered with precaution, since at this stock level, spawning biomass may already be critically reduced. As has been emphasized by Gayanilo and Pauly (1997), in small-sized tropical fish species, *F*-values, which maximize yield per recruit, tend to be high and hence can result in inappropriate management measures. Thus, the recommended reference point $E_{0.1}$ should be the target and this is below current exploitation rate of *L. borbonicus* indicating a slight overexploitation. In contrast, exploitation rates of *L. vaigiensis*, *S. ghobban* and *L. fulviflamma* do not exceed the $E_{0.1}$ reference points, suggesting that these species may be harvested within sustainable limits. Although *L. vaigiensis* is the third most caught species in the catches its exploitation value of 0.56 is comparatively low. This is in contrast to the situation in the southern Kenyan fishery, where exploitation rates are very high (0.7, Hicks and McClanahan, 2012). This might be explained by the length at first capture. In Chwaka Bay L_c-values (17.3) exceed those of the Kenyan fishery (13.6) and are equal to the optimum length at first capture. Overall, analysis, indicate that the general state of the bay`s fishery represents a “full exploitation to slight overexploitation scenario” with no scope for expansion.

3.4.3. Mesh size regulations

The low abundance of large individuals in the catches of the Chwaka Bay fishery might be due to a continuous overexploitation of larger, slower growing specimens in the past, which may have left room for smaller fish to grow and reproduce, shifting the community towards smaller individuals with lower L_∞-values, and higher K-values (Law, 2000). Nonetheless, since large specimens are not totally absent in the catches and maximum length recorded in this study is close to maximum length as reported in FishBase (Froese and Pauly, 2015), this "overexploitation scenario" for large sizes might not be the case for Chwaka Bay. The area is comprised of shallow water habitats that act as nursery and feeding grounds for a large number of target species (Nagelkerken et al., 2000), thus juveniles, or rather small specimens, might occur in higher abundances and larger fish may concentrate further outside the bay area. Large specimens of *S. ghobban* (approximately > 35 cm), for instance, were found to be restricted to reef areas at the outer margins of the study site (Rehren and Wolff, unpublished data). However, as depth and distance from shore increases these reefs are subject to lower fishing efforts. Kimirei et al. (2011) studied the ontogenetic habitat use of, inter alia, *S. sutor*, *L. fulviflamma* and *L. lentjan* in mangroves, seagrass beds, mudflats and reefs in Tanzanian nearshore waters. They found for all four species that adults were mainly present in deep water habitats. This supports the idea of having mainly smaller sized and many juvenile individuals at the main fishing grounds. Under this scenario it is questionable if an increase in mesh size will benefit the fishery by letting the fish grow and spawn, eventually leading to the replenishment of larger size classes. The management of the mesh size in the bay would under the current fishing effort, require an increase in length at first capture by a factor of 1.5 – 3 in order to keep the target species within sustainable limits. Considering a combination of effort and mesh size regulations, length at first capture would still need to be increased by 30 to 40% for *L. lentjan* and *S. sutor*. Such an increase in mesh size would probably lead to a substantial decrease in overall catch volume so that operation costs would exceed yield. Thus, an increase in mesh size requires an extension of the radius of the fishery to capture larger specimens outside the shallow bay area.

3.4.4. Effort reduction and gear change

The current fishing mortality of *S. sutor*, *L. borbonicus* and *L. lentjan* is exceeding sustainable limits implying a need to reduce overall fishing pressure on those target species. Estimates reveal that under current L_c and optimum fishing ($F_{0.1}$) the daily number of boats has to be reduced by 68 % in order to sustainably harvest all species. This would leave roughly 517 fishermen (average 5 fisher $boat^{-1}$) without income.

Consequently, a reduction in effort only would appear as a viable option if income alternatives were available. Seaweed farming and tourism have been proposed but seem to represent insufficient solutions (Eklöf et al., 2012; Gustavsson et al., 2014). In 2014, a persistent conflict upon spatial use and gear utilization between trap and dragnet fisher have led to the attempt by the Department of Fisheries and Marine Resources to actively enforce the prohibition of the use of dragnets within the bay limits. This closure lasted for three weeks and was dropped after massive protest by the suddenly unemployed dragnet fishermen (Pers. comm.). The high number of dragnet fisher (approximately 550 fishermen) and the lack of alternative livelihoods, demonstrates clearly, that this enforcement was condemned to fail from the very beginning. Although traps have shown to lead to higher catches, the use of dragnets in the bay becomes more and more common (de la Torre-Castro and Rönnbäck, 2004). One of the reasons is that fishing with dragnets requires several fishermen to pull the net over the sea floor and hence creates job opportunities for the local community, while trap or handline fisher operate usually with a maximum of 3 people. Furthermore, the lack of financial resources for buying boats, gears and engines pushes many fisher towards joining a dragnet fisher boat (de la Torre-Castro, 2012). A gear exchange program was conducted within the partnership of MACEMP (Partnership initiated in 2006 between the United Republic of Tanzania, the Global Environmental Fund and the World Bank) to provide alternative fishing gears to, inter alia, the dragnet fisher of Chwaka Bay (Gustavsson et al., 2014). However, the low amount of fishing gears provided to the community could not lead to any significant change (Gustavsson, et al., 2014). Indeed, many fishermen argue that the only sustainable solution is the redistribution of dragnet fisher to other more sustainable gears such as traps and handlines. However, if just half of the dragnet fisher would be reallocated to trap and handlines (Average of 1.5 fisher boat^{-1}), this would lead to an increase of 150 boats operating in the bay and may amplify spatial use conflicts as well as generate new problems such as pollution. A redistribution of dragnets to other nets applied in offshore fishing grounds appears as a management measure, which lacks a scientific basis and requires further biological assessments, as it is not clear how productive the offshore fishing grounds on the east coast of Zanzibar are. This clearly shows that a redistribution of dragnet fishermen to other gears cannot be the final solution. The problem of the use of the destructive dragnets and the high fishing mortality can thus only be solved by long-term stepwise reduction in overall effort, particular in dragnet effort.

CHAPTER IV - The Trophic Model of Chwaka Bay

Chapter IV.

Holistic assessment of Chwaka Bay's multigear fishery – using a trophic modelling approach

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Abstract

East African coastal communities highly depend on marine resources for not just income but also protein supply. The multispecies, multigear nature of East African fisheries makes this type of fishery particularly difficult to manage, as there is a trade-off between maximizing total catch from all gears and species and minimizing overfishing of target species and the disintegration of the ecosystem. The use and spatio-temporal overlap of multiple gears in Chwaka Bay (Zanzibar) has led to severe conflicts between fishermen. There is a general concern of overfishing in the bay because of the widespread use of small mesh sizes and destructive gears such as dragnets and spear guns. We constructed an *Ecopath* food-web model to describe the current trophic flow structure and fishing pattern of the bay. Based on this model, we explored the impact of different gears on the ecosystem and the fishing community in order to give advice for gear based management in the bay. Results indicate that Chwaka Bay is a productive, shallow water system, with biomass concentrations around the first and second trophic level. The system is greatly bottom-up driven and dominated by primary producers and invertebrates. The trophic and network indicators as well as the community energetics characterize Chwaka Bay as relatively mature. Traps and dragnets have the strongest impact on the ecosystem and on the catches obtained by other gears. Both gears potentially destabilize the ecosystem by reducing the biomass of top-down controlling key species (including important herbivores of macroalgae). The dragnet fishery is the least profitable, but provides most jobs for the fishing community. Thus, a complete ban of dragnets in the bay would require the provision of alternative livelihoods. Due to the low resource biomass of fish in the bay and the indication of a loss of structural control of certain fish groups, Chwaka Bay does not seem to provide scope for further expansion of the fishery. Instead, we recommend a reduction in the use of dragnets and traps, partially by redistributing them to the more profitable longlines and handlines.

Keywords: Chwaka Bay; small-scale fishery; multigear fishery; gear management; *Ecopath*

4.1. Introduction

Small-scale fishers account for 44 % of fishers worldwide (Teh and Sumaila, 2013) and are often the major livelihood provider for local communities (Walmsley et al., 2006). Particularly, in East Africa coastal communities are highly dependent on marine resources for not just income but also protein supply (van der Elst et al., 2005). This high dependency has led to great pressure on resources and in several instances to the observation of signs of overexploitation of target species and a degradation of ecosystems (Jacquet et al., 2010; McClanahan et al., 2008b; Nordlund et al., 2013). Small-scale fisheries in East Africa are characterized by low-technique gears, wooden boats and the use of multiple gears (Davies et al., 2009; de la Torre-Castro and Rönnbäck, 2004; Mangi and Roberts, 2006). The use of different gears catching a variety of species makes this type of fishery particularly difficult to assess and manage, as there is a trade-off between maximizing total catch from all gears and species and minimizing overfishing of target species and the disintegration of the ecosystem. Marine Protected Areas (MPAs) are considered a prominent tool for sustaining fisheries in the Western Indian Ocean (Wells et al., 2007), but enforcement and community compliance is often weak, which can impede MPA success (Gustavsson et al., 2014; McClanahan et al., 2009b; Rocliffe et al., 2014). Furthermore, there is often a lack of proof that no-take zones indeed benefit coastal communities (Benjaminsen and Bryceson, 2012; Wolff, 2015). McClanahan (2012) showed that in general fishing area management was not well perceived by fishing communities in Kenya. In contrast, gear management, such as mesh size regulation and prohibition of the use of destructive gears, has qualified as the more accepted management measures in East Africa (McClanahan et al., 2012, 2008). Gear management can not only help in progressing towards the sustainable management of fisheries but may also reduce conflicts between fishermen using different fishing methods.

A typical case of a multispecies, multigear fishery with conflicting resource use is Chwaka Bay at the east coast of Zanzibar (Tanzania) (de la Torre-Castro and Lindström, 2010). Here, the community depends highly on its fisheries resources and the fishery is believed to show signs of overexploitation (de la Torre-Castro and Lindström, 2010; de la Torre-Castro et al., 2014), with fishermen reporting a decrease in their catch rates. The widespread use of dragnets[1] in the bay is said to be highly destructive and traditional basket trap fisher complain that by dragging the net over the seafloor, their traps are displaced and destroyed (de la Torre-Castro and Lindström, 2010). In addition, many fisher argue that dragnets sweep up everything in its path, thus taking away the catch for other fishermen using other gears (McClanahan et al., 1997; pers. comm.). It is often stated that the catch per fisher of a dragnet boat is higher than that of other fishers and that dragnets with small mesh sizes yielding large amounts of small fish, may lead to growth overfishing of target resources (de la Torre-Castro et al., 2014; pers. comm.). However, studies from other regions have shown that the more traditional traps and handlines often catch similar fish sizes (Mangi and Roberts, 2006). Furthermore, studies indicate that the bait used by trap fisher leads to large amounts of catches of herbivorous fish (Mbaru and McClanahan, 2013), which could have destabilizing effects on reef systems. Despite the constant concern for Chwaka Bay's fisheries resources, a lack of quantitative assessments on the status of the fishery and gear impacts on the ecosystem, makes it difficult to formulate meaningful management plans.

A useful tool for a holistic description of the various effects of different gears on the ecosystem as well as their downsides and benefits to the fishing community is the trophic modelling software *Ecopath with Ecosim* (Bacalso et al., 2016; Bundy, 1997; Moutopoulos et al., 2013). We used this approach to model the bay`s trophic flow structure and fishing pattern and to explore potential effects of gear choice on the food-web, the key species of the fishery and profits for fishermen.

4.2. Material and Methods

4.2.1. Study area

Chwaka Bay is located on the east coast of Unguja Island, Zanzibar (6°02'6°13'S, 39°24'39°36'E, Fig. 1.1). The climate of the bay is driven by the northeast and southeast monsoon seasons (November to March and April to October, respectively) with the highest precipitation peak occurring during March and a lower second peak during October/November (McClanahan, 1988; Shaghude et al., 2012). The bay is relatively shallow with depth ranging between 3 meter in the bay proper and about 20 meter around the reef at the offshore border. The sea surface temperature has been measured to vary from 25 to 31° C (Jiddawi and Lindström, 2012). Freshwater enters the bay only through rainfall; there is no riverine water input. Salinity ranges from 35 ‰ at the bay opening to about 26 ‰ in the bay proper (Jiddawi and Lindström, 2012). Strong tidal currents, with a mean tidal range of 3.2 meter (Nyandwi and Mwaipopo, 2000) are causing high turbidity in the bay by stirring up sediments (Gullström et al., 2006).

The bay provides three coastal key habitats for nursery and breeding grounds of marine species: dense seagrass meadows, large mangrove stands along the shore and a fringing reef, protecting the bay proper. These distinct habitats are interconnected through the exchange of particulate organic matter (Mohammed et al., 2001) as well as tidal, foraging, seasonal and ontogenetic migration of fish species forming a habitat continuum (Gullström et al., 2012). These characteristics give rise to a very productive and diverse coastal seascape, which drives an intensive nearshore fishery. The local community is highly dependent on fisheries resources for income and protein supply. To secure the conservation of these important resources Chwaka Bay has been made part of the Mnemba Island Marine Conservation Area management plan (MACEMP) since 2005 (McLean et al., 2012). However, the bay remains a general use zone.

The majority of the fishery concentrates around the reefs, covering an area of approximately 124 km^2. It is artisanal, employing wooden boats and low technique gears. Main fishing gears are basket traps, dragnets, handlines, spears and, to a minor extend, longlines, gillnets, floatnets[6] and fence traps. According to Zanzibar's Fisheries Act 2010 the use of dragnets and spear guns is illegal on the Island due to their environmental destructiveness (Jiddawi and Ohman, 2002; Mangi and Roberts, 2006). One of the major fishery targets is the rabbit fish constituting 20 % of the total catch (Principal species *Siganus sutor*). Other important fish are reef associated emperors,

[6] The local name for floatnets is *kokoro* and is referred to in the Frame Survey 2010 (Kathib and Jiddawi, 2010) as surrounding nets.

parrotfish, groupers, wrasses and snappers (28 %) but also pelagic fish such as barracudas, needlefish, halfbeaks, scads and jacks (15 %) are targeted. Two cephalopod species are highly important for the fishery: the octopus *Octopus cyanea* (8.4 %) which is harvested by spear fisher and the reef squid *Sepioteuthis lessoniana* (5.6 %) which is harvested by handline fisher. In addition, foot fisher collect invertebrates in the intertidal zone for both food and income supply (Fröcklin et al., 2014). Large parts of these invertebrate collections are comprised of crabs and gastropods.

4.2.2. The model

The Chwaka Bay food web model was constructed using the software platform *Ecopath with Ecosim* (*EwE*, Version 6.4.3). *Ecopath* composes a mass-balanced steady–state "snapshot" of an ecosystem using a set of linear equations for each of the model compartments that are linked by a diet matrix. The mass-balance principle is simulated through two mayor equations. The first describes the production of functional groups and ensures the energy balance among these different groups:

$$B_i(\frac{P}{B})_i EE_i = Y_i + \sum_j [B_j(\frac{Q}{B})_j DC_{ij}] + BA_i + NM_i \qquad (1)$$

where B_i and B_j are the biomasses of each prey i and predator j, $(P/B)_i$ is the production per biomass ratio, EE_i is the ecotrophic efficiency, which is the part of the total production of each functional group that is used in the system, Y_i is the total yield of the fishery for each functional group, $(Q/B)_j$ is the consumption per biomass ratio of each predator j with DC_{ji} being the contribution of each prey i to the diet of each predator j, BA_i is the biomass accumulation and NM_i the net migration.

And the second equation ensures the mass-balance within each functional group:

$$Q_i = P_i + R_i + GS_iQ_i \qquad (2)$$

where Q_i is the consumption of group i, P_i is the sum of production of group i, R_i is the respiration of group i, and (GS_i x Q_i) is the unassimilated food of group i.

Ecopath is able to solve the above-mentioned equation if three of the following parameters are entered for each functional group: *B*, *P/B*, *Q/B* or *EE* (Christensen et al., 2008).

EE is the proportion of functional group i that is used in the system. An *EE* over 1 indicates that more is being consumed of a functional group than is being produced. If

EE is under 1 a functional group produces more than is directly used in the food web and the remaining amount fuels the detritus pool of the system.

4.2.3. Construction of the food web

The construction of the food web model was primarily based on information of fisheries data collected from January to June and September to December 2014 and former biomass studies conducted in the bay. During the sampling period in 2014, data on species composition (specimens were identified to either species or family level) length, weight and number of species/families were sampled for each fishing gear on 18 days per month at the three mayor landing sites of Chwaka Bay: Chwaka village, Marumbi village and Uroa village (Fig. 4.1). In addition, information on effort and operating costs (fuel, boat and gear costs) was collected. Information about price per species was obtained through the kg price in 2013 as reported by the Department of Marine Fisheries Resources. The following six key species were identified as dominant in the catches: *Siganus sutor* (Valenciennes, 1835), *Lethrinus borbonicus* (Valenciennes, 1830), *Leptoscarus vaigiensis* (Quoy and Gaimaird, 1824), *Lethrinus lentjan* (Lacepède, 1802), *Scarus ghobban* (Forsskål, 1775) and *Lutjanus fulviflamma* (Forsskål, 1775). For these six species detailed information on length frequencies for subsequent stock assessments was collected (Rehren et al., submitted*a*), consequently they were modelled separately. The classification of the other functional fish groups was mainly based on their food preference. Prior to that, species/families were separated into pelagic and non-pelagic fish (A more detailed description of the different functional groups is shown in Table 4.1.). In total, the model is comprised of 11 fish groups, 12 invertebrate groups, 1 zooplankton group and 3 primary producer groups.

Table 4.1. List of the taxa, which contribute to the different functional groups.

#	Group name	Composition
7	Other carnivorous fish	Synodontidae, Platycephalidae, Lutjanidae, Gerreidae, Priacanthidae, Lethrinidae, Labridae, Serranidae, Pleuronectiformes, Hollocentridae, Ostracidae, Haemulidae, Rajiformes, Myliobatiformes, Mullidae, Anguilliformes, Albulidae, Fistulariidae, Plotosidae, Diodontidae, Nemipteridae, Ephippidae, Drepaneidae
8	Pelagic fish	Belonidae, Hemiramphidae, Echeneidae, Coryphaenidae, Chirocentridae, Scombridae, Carangidae, Sphyraenidae, Rachycentridae
9	Other herbivorous fish	Acanthuridae, Scaridae, *Chanos chanos*
10	Zooplanktivorous fish	*Sardinella spp*., Caesionidae, *Siganus stellatus,* Apogoindae, Centriscidae, Engraulidae, Syngnathidae
11	Omnivorous fish	*Lutjanus kasmira*, Kyphosidae, Balistidae, Chaetodontidae, Terapontidae, Mugilidae, *Scarus russelii*, Monodactylidae, Monacanthidae
12	Octopus	*Octopus cyanea*
13	Squids	Mainly *Sepioteuthis lessoniana*. Other: *Sepia latimanus*
14	Crabs and lobsters	Commercial: Portunidae, Palinuridae. Non-commercial: Other brachyuran crabs
15	Other crustaceans	Amphipods, Ostracods, Copepods, Stomatopoda, others
16	Bivalves	Mytilidae, Pinnidae, Mactridae, Cardiidae
17	Gastropods	Commercial: *Lambis spp*., *Chicoreus spp*., Turbinellidae. Others: Ovulidae, Olividae, Trochidae, Potamididae
18	Other echinoderms	Ophiuroidea, Echinoidea, Asteroidea
19	Sea cucumber	*Holothuria atra*, *H. leucospilota*, *H. coluber*, *Actinopyga lecanora*, *A. mauritiania*, *A. echinites*, *Stichopussp.*
20	Annelids	Polychaeta, Oligochaeta
21	Other meiobenthos	Nematoda, Polyplacophora, Sipunculida, Pycnogonida, Branchiostoma, Chironomidae
22	Sessile benthos	Porifera, Tunicata, Actiniaria
24	Corals	Hard and soft corals

4.2.4. Input data: Biomass, P/B, Q/B and diet composition

The biomass estimates of the six key species were derived from Jones cohort analysis (Rehren et al., submitted*a*). For all other fish and invertebrate groups, which are harvested by the fishery, biomass estimates were calculated based on total annual yield and biomass specific productivity assuming an exploitation rate of 0.5. For non-fished invertebrate groups as well as for the producer groups, biomass estimates were taken from published studies conducted within the bay (Eklöf et al., 2005; Gullström et al., 2006; Kyewalyanga, 2002; Sjöö et al., 2011). Insufficient or no biomass estimates were available for *zooplankton*, *corals*, *other meiobenthos* and *sessile benthos* and thus these values were estimated by *EwE* using values of 0.75, 0.6, 0.75 and 0.65 for the ecotrophic efficiency of these groups. *EE* values were estimated based on the relative biomass contribution of these groups to the system and comparable values that have been observed in other similar systems (Bacalso and Wolff, 2014; Chen et al., 2008; Cruz-Escalona et al., 2007; Tsehaye and Nagelkerke, 2008; Vega-cendejas and Arregiun-Sanchez, 2001). Biomass of gastropods and sea urchins was considered underrepresented by the study conducted by Eklöf et al. (2005), because authors were only accounting for macroinvertebrates that were sampled within a PVC-tube (12 cm diameter). Consequently, studies from Lyimo et al. (2008) and Eklöf et al. (2006) were used to complement biomass estimates for the functional groups *gastropods* and *other echinoderms*. The biomass of the detritus group was calculated using the formula provided by Pauly et al. (1993):

$$\log D = -2.41 + 0.954 \log PP + 0.863 \log E \tag{3}$$

Where *PP* is the primary production of the system and *E* is the euphotic zone.

As the *P/B* ratio is equal to the total mortality (*Z*) under steady state conditions (Allen, 1971), *Z*-values derived from catch curve analysis were used for biomass specific production of the six key species. For all other fish groups, *Z* (*P/B*) was taken from stock assessment analysis conducted in the region or from other similar *Ecopath* models. For the invertebrate groups and the zooplankton group *P/B* values were mainly obtained from similar models. For primary producers studies from the bay were available (Kyewalyanga, 2002; Lyimo et al., 2006), except for macroalgae, for which a study from Brazil was used (Freire et al., 2008).

The empirical equation of Palomares and Pauly (1998) was used to calculate *Q/B* values:

$$\log\left(\frac{Q}{B}\right) = 7.964 - 1.965T - 0.2041 logW_{\infty} + 0.083A + 0.532h + 0.398d \tag{4}$$

where *T* is the water temperature, W_{∞} is the asymptotic body weight, h is the food type (0 for herbivores and 1 for predators) and A is the aspect ratio defined as $A=h^2/s$, with h being the height of the caudal fin and s its surface area. W_∞ was converted from L_∞-values using the *a* and *b*-values from length-weight relationships. L_∞, *a* and *b* were obtained from FishBase (Froese and Pauly 2015). For several species the aspect ratio was measured during the sampling period in 2014. For other fish groups, information on the aspect ratio was obtained from FishBase (Froese and Pauly 2015). If the aspect ratio was unknown, values for *Q/B* were taken from other models. Likewise, Q/B ratios for invertebrate groups were mainly obtained from other *Ecopath* models of similar systems.

Diet information was taken from stomach content and stable isotope studies conducted in Chwaka Bay or within the region, which was available for the majority of the fish groups. For the rest of the fish groups as well as the invertebrate groups diet information was taken from studies conducted in similar regions or FishBase (Froese and Pauly 2015). The basic input parameters and the diet matrix for the *Ecopath* model are shown in Table 4.2. and 4.3. (For detailed information on data sources and initial parameters see Fig. S.4.1 and Fig. S.4.2.). In order to evaluate data quality, the pedigree routine was used, which quantifies the uncertainty of the input parameter (Christensen et al., 2008). Each input parameter is assigned a value from 0 to 1 (0 = low quality data, 1 = high quality data) and subsequently, *Ecopath* calculates an overall average of pedigree from all input data.

4.2.5. Model balancing

The *Ecopath* model is based on the assumption of mass balance, a situation, which is only given if for all groups the *ecotrophic efficiency* is below 1. If the computed values are > 1, biomass estimates and/or P/B ratios are too low to account for all the assumed predation/fishery in the model. In our model we adjusted the *EE*-values by modifying the diet composition of the non-fished and fished groups that we considered the most unreliable, since diet information for these groups was partly missing in the region and needed to be taken from other studies/models of similar systems. Another criterion used to balance the model was the gross efficiency (production/consumption, P/Q) for each functional group that usually ranges between 0.1 and 0.3 (Christensen et al., 2008). Accordingly, Q/B values of *other crustaceans*, *Lethrinus lentjan* and *Lutjanus fulviflamma* were adjusted to fall into this range.

Since, several of the input parameters show a relatively high degree of uncertainty (Particularly, parameters that were obtained from other models or similar systems), we used the *Ecoranger* routine of the *Ecopath with Ecosim* Version 5 (Pauly and Christensen, 1993) to create alternative balanced versions of the model. This resampling routine allows defining mean and ranging values for all input parameters to account for their uncertainty and then draws random input parameter in a Monte-Carlo fashion. The resulting models are evaluated based on user defined criteria, physiological and mass-balance constraints (EE < 1, P/Q between 0.1 - 0.3). We let *Ecoranger* run 10000 times using the unbalanced state of the model and from the model runs that passed the selection criteria the best fitting ones were chosen (least sum of squared residuals). Results of the MTI routine mean trophic level of the catch of the different gears and summary statistics for the different alternative models were than compared with the Chwaka Bay reference model presented here. None of the alternative models yielded qualitative different results and since the ecotrophic efficiencies of the different groups in our original model were considered ecologically more coherent than in the alternative models, it was chosen for further analysis. In addition we used *Ecoranger* to perform a simple sensitivity routine by altering the input parameter of the different functional groups stepwise from -50 % to 50 % and checking what effect this change has on all of the “missing” basic parameters (Christensen et al., 2005).

4.2.6. Indicators used to characterize the Chwaka Bay food-web

The catch per unit of area of the fishery (t km^{-2} yr^{-1}, CPUA) and the gross efficiency of the catch (catch/primary production, GE) were used to describe the magnitude and energetic efficiency of the Chwaka bay fishery. Several ecological indices calculated by the model (Table 4.4.) were used to describe the state and size of the Chwaka Bay ecosystem and to measure the efficiency of energy transfer and amount of cycling in the system (For detailed description of these indicators see Bacalso and Wolff, 2014; Christensen, 1995). In addition, the lindeman spine flow diagram was used to visualize the different energy flows between the aggregated discrete trophic levels of the food-web.

4.2.7. Assessment of the gear impact on the fisheries resources

For each fished group/species, its mean trophic level is calculated in *Ecopath* as the weighted mean of the trophic level of its prey groups +1 (Christensen et al., 2005). By calculating the contribution of the different gears to the groups`s catch it was then possible to compute the mean trophic level of the catches for all single gears. As a default, *Ecopath* also calculates the mean trophic level of the entire catch of all gears, which is calculated as the weighted mean of the trophic level of each target group (Christensen et al., 2005). As an index of species diversity in the catches of the different gears, the Shannon-Wiener Index was used and the respective standard deviation calculated for each fishing gear using all the sampled fishing trips in 2014. Estimates were based on weight (kg) per species.

To obtain a measure of size selectivity, the mean length and the standard deviation of the catch of the 15 dominant fish families was estimated for dragnets, traps, handlines, floatnets, and spears, for which sufficient samples were available. To test for differences in the mean length of the catch of these gears we used a one-way ANOVA with a Tukey's post hoc test.

4.2.8. Mixed trophic impact (MTI) and economic analysis

We used the mixed trophic impact routine of *Ecopath* to assess how a small increase in catch of one gear impacts the catch of the other gears as well as the biomass of the functional groups. This routine quantifies the impact by using a range from 1 (positive impact) to -1 (negative impact).

In addition, we compared the overall total profit of each gear, the catch per unit of effort (Catch fisher^{-1} fishing trip^{-1}), the individual profit (Profit fisher^{-1} fishing trip^{-1}) and the number of fisher employed by the different gears. Profits are given in TZS (1 USD = 1663.2 TZS, 2014). In *Ecopath* profits are calculated by multiplying the market price (TZS) of each functional group by its catch (t km^{-2} yr^{-1}) and subtracting all user defined costs (i.e. fixed costs, effort related costs, sailing related costs). In the Chwaka Bay model we only defined two cost categories, sailing related costs and fixed costs. The sailing related costs are defined as the total fuel cost of fishing boats of each gear per year. To calculate total fuel costs we used an average fuel cost per boat trip and multiplied this by the numbers of boats using engines times their trips per year. Fixed costs are comprised of a) gear costs, which vary according to the gear in use b) an average boat cost multiplied by the number of boats of each gear and c) an average engine cost multiplied by the amount of engines used by the different gears.

4.3. Results

Table 4.2. Input parameters of the Chwaka Bay food-web model and the ecotrophic efficiencies (EE) as well as production over consumption ratios (P/Q) of the different functional groups. Values in brackets were calculated by Ecopath.

	Group name	TL	Habitat area	Biomass (t/km²)	Production / biomass (/year)	Consumption / biomass (/year)	Ecotrophic efficiency	Production / consumption
1	*Siganus sutor*	(2.11)	1	0.524	3.73	26.60	(0.92)	(0.14)
2	*Leptoscarus vaigiensis*	(2.03)	1	0.174	2.09	20.39	(0.91)	(0.10)
3	*Lethrinus lentjan*	(3.28)	1	0.141	2.64	18.00	(0.93)	(0.15)
4	*Lethrinus borbonicus*	(3.35)	1	0.090	3.56	24.79	(0.92)	(0.14)
5	*Lutjanus fulviflamma*	(3.48)	1	0.128	2.12	17.00	(0.94)	(0.12)
6	*Scarus ghobban*	(2.15)	1	0.328	1.07	20.00	(0.93)	(0.05)
7	Other carnivorous fish	(3.36)	1	0.931	2.19	8.72	(0.92)	(0.25)
8	Pelagic fish	(3.24)	1	0.553	2.16	12.14	(0.86)	(0.18)
9	Other herbivorous fish	(2.04)	1	0.053	3.32	32.75	(0.94)	(0.10)
10	Zooplanktivorous fish	(2.86)	1	0.115	3.53	15.35	(0.92)	(0.23)
11	Omnivorous fish	(2.39)	1	0.114	2.87	10.53	(0.91)	(0.27)
12	Octopus	(3.54)	1	0.200	4.00	16.00	(0.83)	(0.25)
13	Squids	(3.25)	1	0.148	3.64	16.60	(0.85)	(0.22)
14	Crabs and lobsters	(2.78)	1	4.126	5.05	22.00	(0.64)	(0.23)
15	Other crustaceans	(2.30)	1	5.880	15.75	52.51	(0.75)	(0.30)
16	Bivalves	(2.15)	1	5.818	1.84	9.58	(0.80)	(0.19)
17	Gastropods	(2.12)	1	5.299	3.52	12.75	(0.79)	(0.28)
18	Other echinoderms	(2.19)	1	10.491	1.24	4.95	(0.47)	(0.25)
19	Sea cucumber	(2.00)	1	0.038	4.45	17.80	(0.82)	(0.25)
20	Annelids	(2.35)	1	11.429	4.50	22.50	(0.89)	(0.20)
21	Other meiobenthos	(2.12)	1	(6.114)	8.55	34.20	0.75	(0.25)
22	Sessile benthos	(2.00)	1	(21.598)	2.00	14.01	0.65	(0.14)
23	Zooplankton	(2.02)	1	(1.799)	40.00	(142.86)	0.75	0.28
24	Corals	(2.10)	1	(5.886)	2.30	7.15	0.60	(0.32)
25	Phytoplankton	(1.00)	1	17.186	82.24	0.00	(0.46)	
26	Macroalgae	(1.00)	0.20	41.200	13.25	0.00	(0.46)	
27	Seagrass	(1.00)	0.25	125.250	3.95	0.00	(0.09)	
28	Detritus	(1.00)	1	51.940			(0.24)	

Table 4.3. Diet matrix of the Chwaka Bay food-web model (Part 1).

	Prey \ predator	1	2	3	4	5	6	7	8	9	10	11	12
1	*Siganus sutor*	0	0	0	0	0	0	0.025	0.075	0	0	0	0
2	*Leptoscarus vaigiensis*	0	0	0	0	0	0	0.0015	0.009	0	0	0	0
3	*Lethrinus lentjan*	0	0	0.005	0	0	0	0.008	0.006	0	0	0	0.015
4	*Lethrinus borbonicus*	0	0	0.009	0	0.007	0	0.005	0.006	0	0	0.001	0.0002
5	*Lutjanus fulviflamma*	0	0	0.007	0	0.003	0	0.006	0.006	0	0	0	0.0005
6	*Scarus ghobban*	0	0	0	0	0	0	0.006	0.008	0	0	0	0.020
7	Other carnivorous fish	0	0	0.075	0	0.030	0	0.013	0.030	0	0	0.004	0.007
8	Pelagic fish	0	0	0	0	0	0	0.020	0.022	0	0	0	0
9	Other herbivorous fish	0	0	0	0	0	0	0.005	0.006	0	0	0.001	0
10	Zooplanktivorous fish	0	0	0.004	0	0.003	0	0.009	0.026	0	0	0	0
11	Omnivorous fish	0	0	0.009	0	0.020	0	0.013	0.005	0	0	0	0.005
12	Octopus	0	0	0	0	0	0	0.003	0.004	0	0	0	0.000
13	Squids	0	0	0	0	0	0	0.004	0.005	0	0	0	0
14	Crabs and lobsters	0	0	0.300	0.200	0.300	0	0.240	0.045	0	0.042	0.019	0.550
15	Other crustaceans	0.021	0.022	0.180	0.560	0.570	0	0.240	0.230	0	0.200	0.040	0.172
16	Bivalves	0	0	0.070	0.100	0.008	0	0.050	0.001	0	0.008	0.007	0.090
17	Gastropods	0.002	0	0.111	0.140	0.020	0	0.100	0.010	0	0.006	0.015	0.140
18	Other ecinoderms	0	0	0.048	0	0.008	0	0.070	0	0	0	0.018	0.000
19	Sea cucumber	0	0	0	0	0	0	0.000	0	0	0	0.0002	0
20	Annelids	0	0	0.006	0	0.030	0	0.078	0.005	0	0	0.010	0
21	Other meiobenthos	0	0	0	0	0	0	0.014	0.012	0	0	0.050	0
22	Sessile benthos	0.084	0	0	0	0	0	0.010	0	0.011	0	0.022	0
23	Zooplankton	0.001	0	0	0	0	0	0.019	0.243	0	0.500	0.120	0
24	Corals	0	0	0	0	0	0.132	0.011	0	0.025	0	0.030	0
25	Phytoplankton	0	0	0	0	0	0	0	0.001	0.012	0.120	0.053	0
26	Macroalgae	0.813	0.006	0.072	0	0	0.760	0.005	0	0.728	0	0.186	0
27	Seagrass	0.034	0.954	0	0	0	0	0	0	0.029	0	0	0
28	Detritus	0.045	0.018	0.104	0	0.0010	0.108	0.046	0.043	0.195	0.124	0.424	0
	Import	0	0	0	0	0	0	0	0.200	0	0	0	0
	Sum	1	1	1	1	1	1	1	1	1	1	1	1

Table 4.3. Diet matrix of the Chwaka Bay food-web model (Part 2).

	Prey \ predator	1	2	3	4	5	6	7	8	9	10	11	12
1	*Siganus sutor*	0	0	0	0	0	0	0.025	0.075	0	0	0	0
2	*Leptoscarus vaigiensis*	0	0	0	0	0	0	0.0015	0.009	0	0	0	0
3	*Lethrinus lentjan*	0	0	0.005	0	0	0	0.008	0.006	0	0	0	0.015
4	*Lethrinus borbonicus*	0	0	0.009	0	0.007	0	0.005	0.006	0	0	0.001	0.0002
5	*Lutjanus fulviflamma*	0	0	0.007	0	0.003	0	0.006	0.006	0	0	0	0.0005
6	*Scarus ghobban*	0	0	0	0	0	0	0.006	0.008	0	0	0	0.020
7	Other carnivorous fish	0	0	0.075	0	0.030	0	0.013	0.030	0	0	0.004	0.007
8	Pelagic fish	0	0	0	0	0	0	0.020	0.022	0	0	0	0
9	Other herbivorous fish	0	0	0	0	0	0	0.005	0.006	0	0	0.001	0
10	Zooplanktivorous fish	0	0	0.004	0	0.003	0	0.009	0.026	0	0	0	0
11	Omnivorous fish	0	0	0.009	0	0.020	0	0.013	0.005	0	0	0	0.005
12	Octopus	0	0	0	0	0	0	0.003	0.004	0	0	0	0.000
13	Squids	0	0	0	0	0	0	0.004	0.005	0	0	0	0
14	Crabs and lobsters	0	0	0.300	0.200	0.300	0	0.240	0.045	0	0.042	0.019	0.550
15	Other crustaceans	0.021	0.022	0.180	0.560	0.570	0	0.240	0.230	0	0.200	0.040	0.172
16	Bivalves	0	0	0.070	0.100	0.008	0	0.050	0.001	0	0.008	0.007	0.090
17	Gastropods	0.002	0	0.111	0.140	0.020	0	0.100	0.010	0	0.006	0.015	0.140
18	Other ecinoderms	0	0	0.048	0	0.008	0	0.070	0	0	0	0.018	0.000
19	Sea cucumber	0	0	0	0	0	0	0.000	0	0	0	0.0002	0
20	Annelids	0	0	0.006	0	0.030	0	0.078	0.005	0	0	0.010	0
21	Other meiobenthos	0	0	0	0	0	0	0.014	0.012	0	0	0.050	0
22	Sessile benthos	0.084	0	0	0	0	0	0.010	0	0.011	0	0.022	0
23	Zooplankton	0.001	0	0	0	0	0	0.019	0.243	0	0.500	0.120	0
24	Corals	0	0	0	0	0	0.132	0.011	0	0.025	0	0.030	0
25	Phytoplankton	0	0	0	0	0	0	0	0.001	0.012	0.120	0.053	0
26	Macroalgae	0.813	0.006	0.072	0	0	0.760	0.005	0	0.728	0	0.186	0
27	Seagrass	0.034	0.954	0	0	0	0	0	0	0.029	0	0	0
28	Detritus	0.045	0.018	0.104	0	0.0010	0.108	0.046	0.043	0.195	0.124	0.424	0
	Import	0	0	0	0	0	0	0	0.200	0	0	0	0
	Sum	1	1	1	1	1	1	1	1	1	1	1	1

4.3.1. The Chwaka Bay trophic model

Table 4.4. Summary statistics for the Chwaka Bay model.

Parameter	Chwaka Bay
Sum of all consumption	1700
Sum of all exports	1493
Sum of all respiratory flows	962
Sum of all flows into detritus	1957
Total system throughput	6112
Sum of all production	2851
Mean trophic level of the catch	2.83
Gross efficiency (catch/net p.p.)	0.00195
Total primary production/total respiration	2.55
Total primary production/total biomass	9.24
Total biomass/total throughput	0.043
Total biomass (excluding detritus)	266
Total catch	4.77
Connectance index	0.32
System omnivory index	0.17
Transfer efficiencies	13.8
Finn's cycling index	3.69
Finn’s mean path length	2.49
Overhead	75

The trophic model of Chwaka Bay (Fig.4.2.) is comprised of 28 functional groups (Table 4.2.) and has a comparatively high overall pedigree of 0.53 (Morissette, 2007), due to high quality and precision of (mostly locally-derived) input data. The sensitivity analysis revealed that changes in input parameters of one functional group (from -50 % to + 50 %) had generally small effects on “missing” parameters of other functional groups indicating relative model robustness. Only the consumption rates of p*elagic fish*, *crabs and lobsters*, *other crustaceans,* g*astropods* and *annelids* strongly affected (> 20 %) the “missing” B values of nine other functional groups (see Fig. S.4.3.).

The food web can be aggregated into 4 discrete trophic levels with 57.8 % of the biomass contributed by primary producers. Of the consumer biomass, 92.2 % is concentrated at the second trophic level, with invertebrate groups being the most dominant (98.4 %). The third trophic level is dominated by *crabs and lobsters* (64.2 %). Most of the fish biomass (63.2 %) occurs at the third trophic level. The lindeman spine diagram (Fig. 4.3.) shows that the highest total system throughput is generated by the primary producers and the second trophic level (63.8 %). Primary producers also contribute most to the flow into detritus (77.5 %). Less than 5 % of the total system throughput is produced by the trophic levels above two.

Selected summary statistics are presented in Table 4.4. and show the community energetics as well as some trophic, fishery and network indicators of the Chwaka Bay ecosystem model. The TST is 6112 t km^{-2} yr^{-1} of which 38 % is consumed, 15 % is exported out of the system, 26 % flows into the detritus and respiration contributes only 22 % to the overall flows. PP/R, PP/B, B/TT values are 2.5, 9.24 and 0.043, respectively. Mean transfer efficiency between trophic levels is 13.8 % and Finn's cycling index and mean path length is 3.69 and 2.49. The system overhead is 75 % and accordingly the ascendency is 25 %. The mean trophic level (TL) and the gross efficiency (GE) of the catch are 2.83 and 0.0019, respectively.

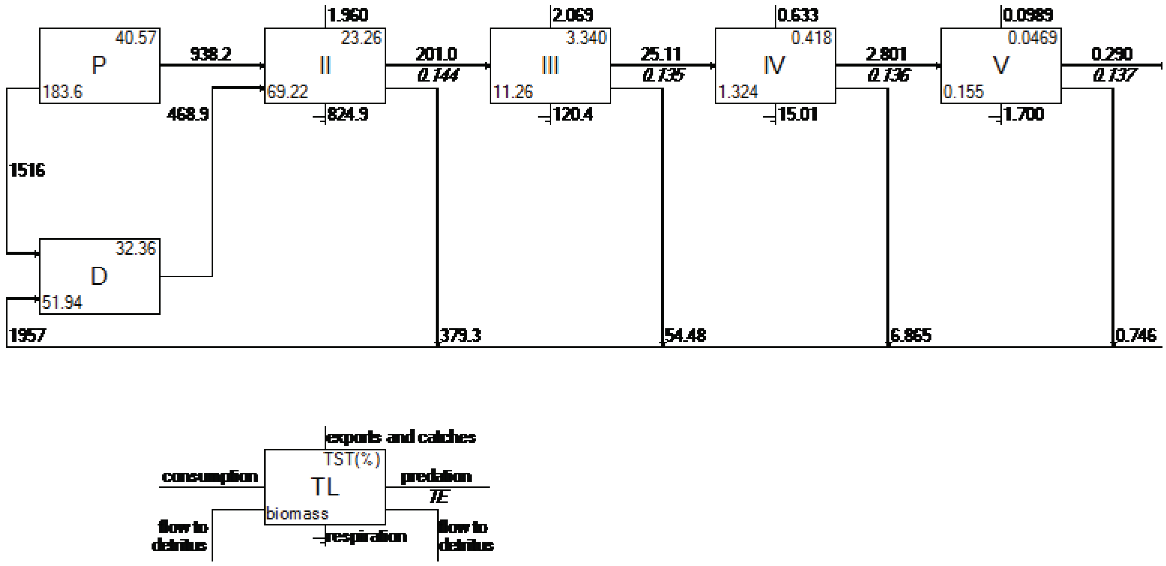

Fig. 4.1. Lindeman spine visualization of the trophic flows up to the fifth trophic level of the Chwaka Bay food-web model. System flow network is aggregated into discrete trophic levels. Flow rates are expressed in t/km^2.

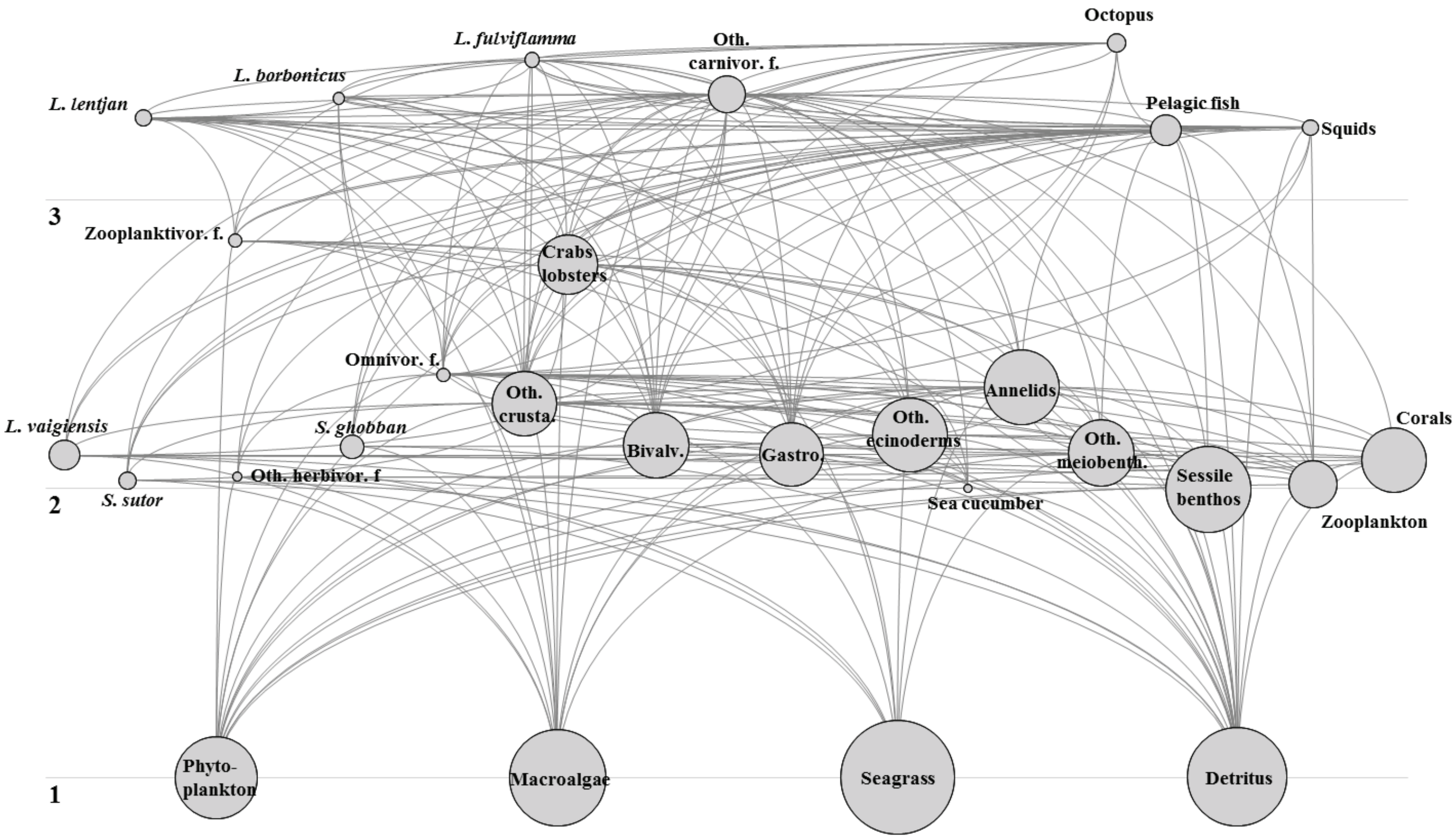

Fig. 4.2. Flow diagram of the Chwaka Bay ecosystem organized by trophic levels. The circles are proportional to the biomass of each compartment and the arrows represent the flows between the different compartments.

4.3.2. Ecosystem impacts of the different gears

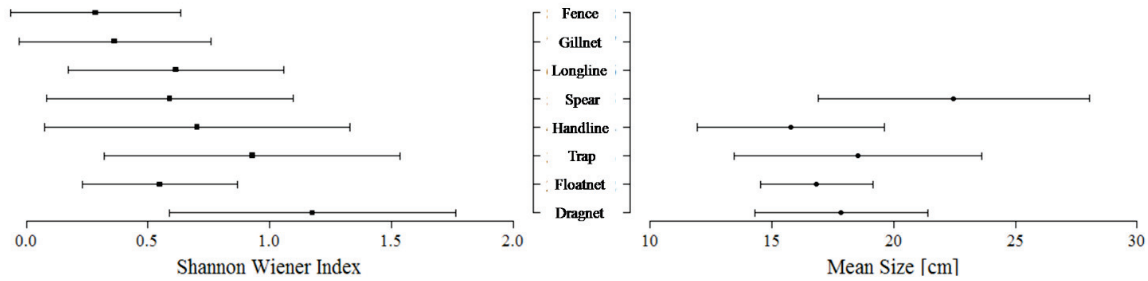

Fig. 4.3. Shannon-Wiener Index of the catch composition for all gears and the mean size distribution of the catch of 15 families from other carnivorous fish, other herbivorous fish and the 6 key species for dragnet, trap, handline, spear and floatnet.

Catches of gillnets, handlines and floatnets showed the highest mean trophic levels (3.31, 3.22 and 3.23, respectively), followed by spears, fence and longlines (3.1, 3.16 and 2.97, respectively). In contrast traps target species from lower trophic levels such as herbivorous fish and thus their catches show the lowest overall trophic level (2.67). The catch of dragnets shows a higher TL then that of traps but is lower than for the rest of the gears (2.88). Fig. 4.4. shows the impacts of the different gears on selected, fished groups. Traps and dragnets have the widest negative impact on Chwaka Bay's food-web with dragnets having a more evenly distributed impact, while the impact of traps is highest on herbivorous fish. Traps also have the strongest impact on the three most important fish species of the bay's catch (*S. sutor*, *L. vaigiensis*, *L. lentjan*). In contrast, spears and handlines show relatively small negative impacts on fished groups, except for *octopus* and *squids* as well as the key species *L. borbonicus*. These two gears also induce small positive effects on pelagic fish and together with floatnets they positively impact several of the key species. Dragnets also have the greatest overall negative impact on the other gears, with its highest impact on the catch of floatnets and fences. Highest impacts of traps are on the catch of dragnets, but not exceeding the impact dragnets have on traps. Spear and floatnet are the only gears that have a positive effect on other gears. As shown by Fig. 4.5., the mean size of the catch differs significantly between all gears except for traps and dragnets (Tukey's test $p < 0.05$). Handlines catch the smallest sizes, followed by floatnets and spears catch the largest specimens. As seen by the Shannon-Wiener-Index of catch diversity (Fig. 4.5.) floatnet and spears represent the most selective fishing methods, while dragnets, traps and handlines are less selective with dragnets being the most unselective.

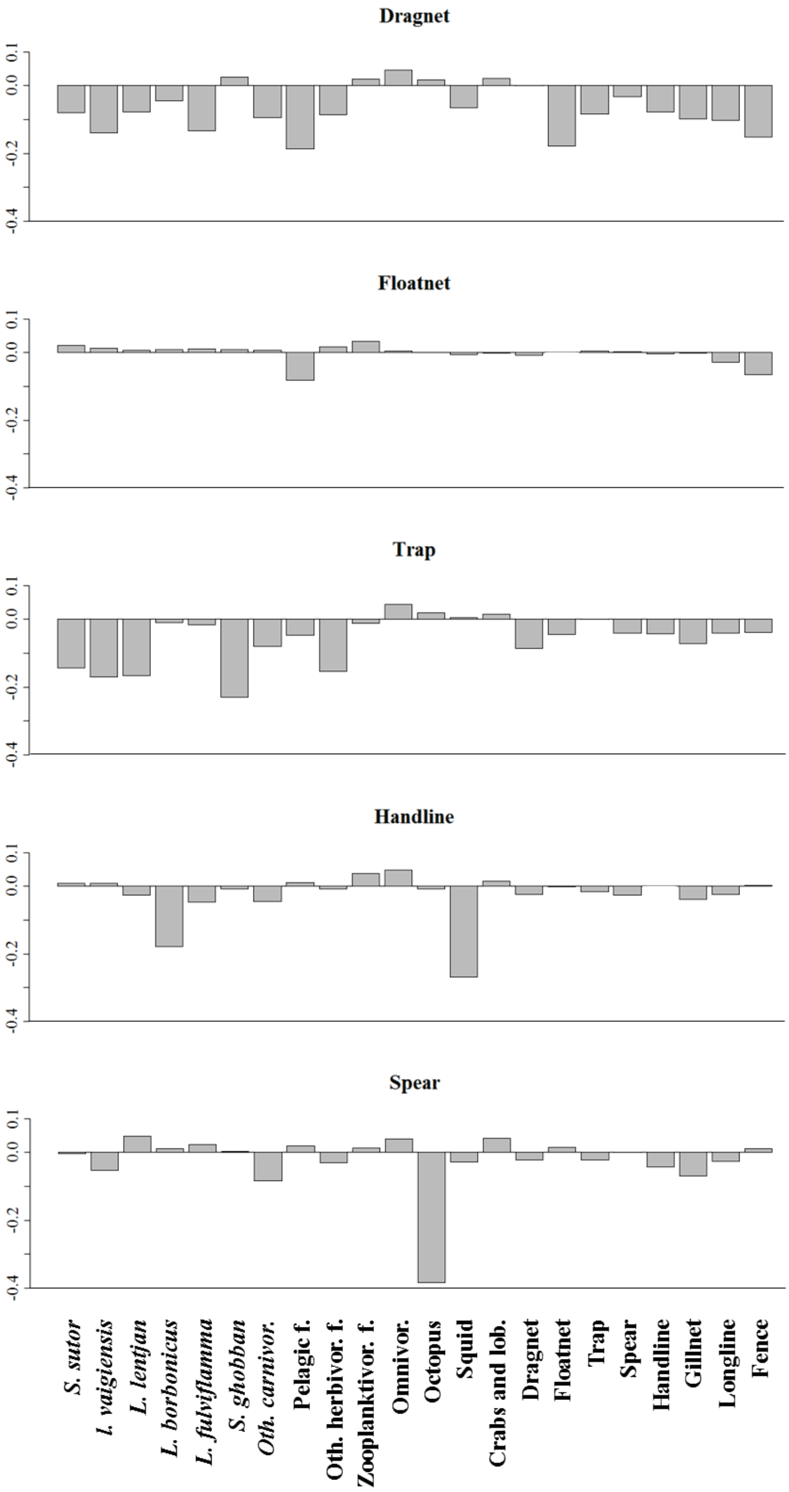

Fig. 4.4. Mixed trophic impact routine, showing the effects of the different gears upon each other and on selected functional groups (Excluded are the impacts a gear has on itself).

4.3.2. Economic analysis

Fig. 4.6. depicts total catch (t km^2 yr^{-1}), CPUE (t fisher^{-1} yr^{-1}), total number of fisher, overall profit (TZS gear^{-1} yr^{-1}) and individual profit (TZS fisher^{-1} fishing trip^{-1}). The number of operating dragnet fisher is three times greater than that of trap fisher, however, trap fisher obtain a similar annual yield. For spear and handlines the number of fisher is similar, but spears have a higher annual catch per area. Traps and dragnets generate also the highest annual profit, twice as much as spears and handlines. There are very few floatnet, longline, fence and gillnet fisher contributing comparatively little to the total catch of the bay and thus to the overall profit. However, longline fisher produce the highest CPUE and individual profit. Dragnet and floatnets provide the lowest profit of all gears.

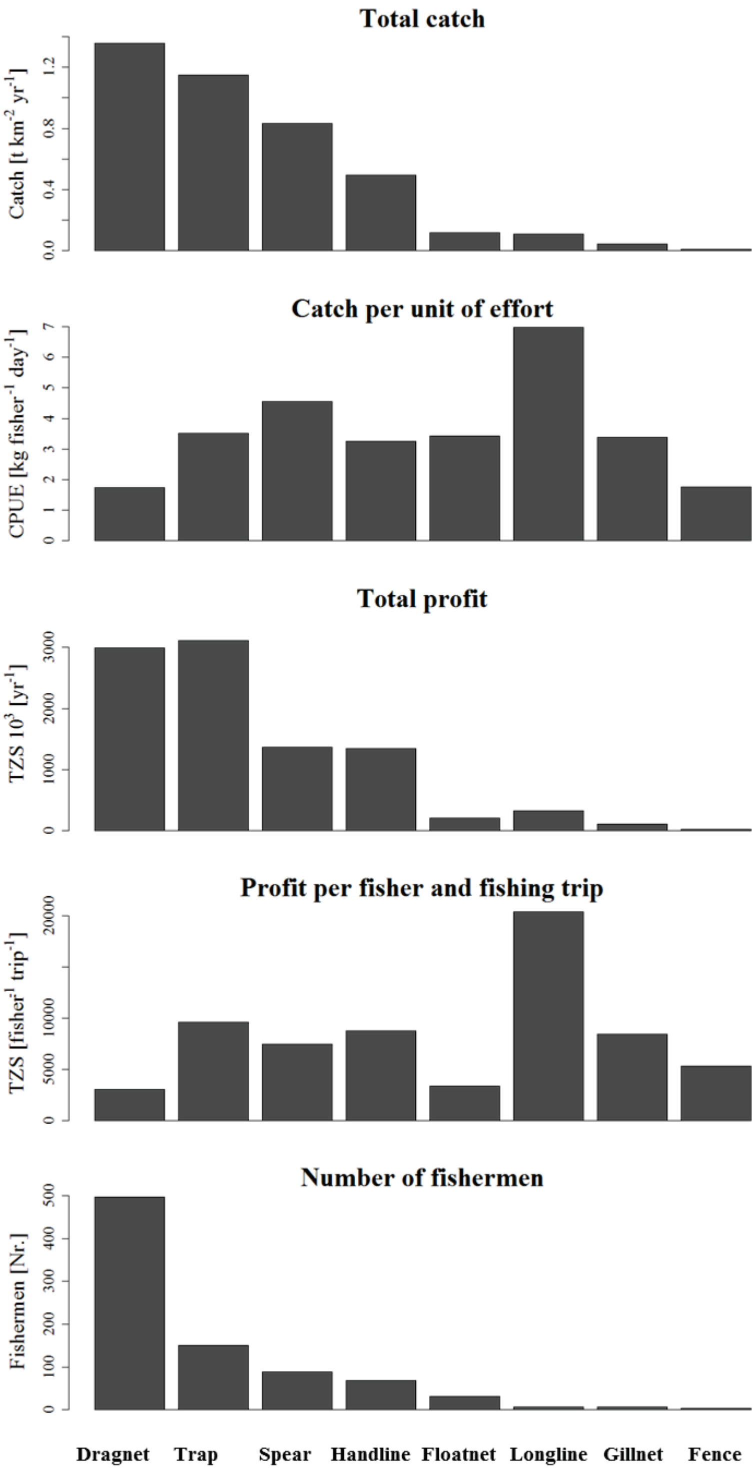

Fig. 4.5. A comparison of the total catch (t km^{-2} yr^{-1}), catch per unit of effort (kg fisher^{-1} day^{-1}), total profit (TZS 10^{3} yr^{-1}), the profit per fishermen (TZS fisher^{-1} trip^{-1}) and an estimated number of fisher for dragnet, trap, spear, handline, floatnet, longline, gillnet and fence respectively.

4.4. Discussion

4.4.1. General characteristics of the Chwaka Bay food-web and the fishery

Chwaka bay is a typical shallow water coastal ecosystem, which is greatly bottom-up driven, due to high macrophyte biomass and benthic production. The biomass of consumers concentrates around the second and third trophic level and is dominated by invertebrates. The system characteristics of Chwaka bay (Table 4.4.) compare well with other bay system models and in general with other models from the Indian Ocean region (Heymans et al., 2014). When judging from the high system overhead, the descriptors of community energetics (B/TST, PP/B and PP/R) and the relatively high connectance index, the system appears as relatively mature and thus little disturbed (Ulanowicz, 1986). In particular, the low PP/R ratio and the high B/TST are similar to other as mature characterized systems (Albouy et al., 2010; Bacalso and Wolff, 2014; Moutopoulos et al., 2013). Since strong disturbances, including fisheries can reduce the stability and maturity of an ecosystem (Gascuel et al., 2008; Pauly et al., 2000), it seems that the fishery in Chwaka Bay has not yet pushed the system back to a more immature state. However, without knowing earlier stages of the Chwaka Bay ecosystem these conclusions remain speculative. On the other hand, indicative for a strong fishing pressure is the overall low biomass of the fish groups, when compared with other models (Bacalso and Wolff, 2014; Cáceres et al., 2016; Cruz-Escalona et al., 2007; Liu et al., 2009). Further, the transfer efficiencies of a food-web can also be used to detect impacts of the fishery on the system, since the fishery is treated as an additional predator for its calculation and an increase in fishing pressure leads to higher efficiencies (Gascuel et al., 2008). The relative high transfer efficiency (13.8 TE) of the Chwaka Bay food-web model when compared with the literature (Chen et al., 2015; Cruz-Escalona et al., 2007; Lin et al., 2004; Liu et al., 2009) might be a reflection of the bay`s strong fisheries exploitation.

Our catch per unit of area estimate (4.77 t km^{-2} yr^{-1}) compares well with estimates from Southern Kenyan fisheries (ranging from 3.36 - 6.6 t km^{2} yr^{-1}, Samoilys et al., 2017; McClanahan 2001), which are very similar in terms of catch composition, gear and boat use (Fulanda et al., 2011; McClanahan and Mangi, 2004). Since our catch estimate does only include the foot collector catch, reported at the landing sites, while it is known that a large part of foot fisher does not sell their catch, the actual CPUA of invertebrates may thus be much higher than estimated in this study. The collection of invertebrates is a major source of protein supply for the local people and shells are used as souvenirs by tourists (Gössling et al., 2004; Nordlund et al., 2010). While a decline of invertebrate populations has been reported (Fröcklin et al., 2014), no study is available that has quantified or qualified the status of these important resources. This is critical as the Chwaka Bay food-web is strongly dominated by invertebrate groups and it highlights the need for their assessment, particular in order to arrive at realistic catch estimates.

The habitat diversity and complexity together with the richness of fisheries resources makes Chwaka bay a very important fishing area. Moreover, fishing in the bay is less influenced by the monsoon seasons than in other areas, due to the bay`s protection by its fringing reef, which allows for a year-round activity with relatively stable catches (de la Torre-Castro et al., 2014). In other parts of Zanzibar and East Africa, as a response to rough wind conditions, many fishermen migrate to other fishing grounds to maintain their catches (Wanyonyi et al., 2016; Mildenberger et al., 2015).

In addition, the shallow waters of the bay facilitate the use of dragnets, which is a highly labour intensive fishery that provides a great number of jobs. Accordingly, this fishery offers income for the majority of the community (Jiddawi, 2012) and the density of fisher in Chwaka Bay (approx. 7 fisher km^2) is relatively high compared to other East African fisheries (Diani-Chale, Kenya 4 fisher km^2, Samoilys et al., 2017).

4.4.2. Ecological and economic impacts of the gears

Traps and dragnets are the most important gears of the Chwaka Bay fishery, generating 53 % of the total annual catch. They also generate the highest overall profit. In contrast, the low effort gears (i.e. longlines, gillnets, floatnets and fence) contribute very little to the overall catch and profit of Chwaka Bay’s fishery. Interestingly, longlines are the most profitable gears exceeding catch per unit of effort and individual profit of the main gears on average by a factor of two. Nevertheless, since our economic analysis does not account for spatial differences in fishing grounds and travel costs, these estimates need to be considered as preliminary. Further, we did not differentiate between fishermen and the captain, who usually owns the equipment and as such bears all the costs and gets a higher share (de la Torre-Castro and Rönnbäck, 2004). Gillnets generate profits for fishermen that are similar to handlines, spears and traps, which has also been observed in the Kenyan fishery (Mangi et al., 2007). Dragnets are the least profitable gear, as they generate only one third as much profit as the other main gears. The low profitability of dragnets agrees with beach seines profits estimated for the Kenyan fishery (Mangi et al., 2007). In Chwaka Bay the catch per dragnet boat is on average divided by 9 fisher and handline and trap boats operate with only 2-3 fisher. This is due to the labour-intensive character of the fishery, since the net has to be pulled over the seafloor and fish has to be driven into the net. Because a dragnet fishermen does not need financial means to buy any equipment, dragnets represent an easy entrance into the fishery and there is a steady increase in their use (de la Torre-Castro and Rönnbäck, 2004). This has already led to a very high number of dragnet fishermen operating in the bay. Consequently, this gear represents an important source of livelihood providing income to a large part of the community and cannot easily be replaced.

Our analysis also demonstrates that dragnets, which, due to their large spectrum of fish species caught, would strongly impact the catch of all other gears if effort was increased. Despite, having a similar species diversity and mean trophic level of catch as trap and handline fisher, dragnets have the highest impacts on floatnets, longlines, gillnets and fence fisher. These effects are mainly driven by the competition for *pelagics* and *other carnivorous fish*. A positive effect of spear fisher on the catch of fence and floatnet fisher and vice versa indicates that such a combination of gears could be favourable, due to the reduction in predation and competition on target species. These gears also have positive effects on several of the key species, heavily caught by dragnets and traps. Furthermore, the spear fishery is much more selective than the other main gears (i.e. handline, dragnet and trap) as their catch is less divers and the overall mean fish length is significantly higher. Dragnets, traps and handlines have a very small mean length of the catch and are highly unselective, targeting a wide range of species from high to low trophic level fish. However, they also exploit some of the more productive and less vulnerable small and fast-growing species (e.g. *Gerres longirostris* and *Octopus cyanea*, (Grandcourt et al., 2006; Guard and Mgaya, 2002), which are often comparatively resilient to overfishing (Jennings et al., 1998; Reynolds et al., 2005).

It has been shown that the seagrass areas of Chwaka Bay are densely inhabited by juveniles and that the abundance of adult specimens within the bay is low (Gullström et al., 2008). Accordingly, larger specimens of reef-associated carnivores such as groupers and emperors are only caught in deeper areas of the bay, where the fishing effort decreases, as fishing gets more difficult. Thus, the choice of gear and mesh size in Chwaka Bay might reflect the natural occurrence of low trophic level fish and high densities of juvenile fish due to the habitat characteristics. Furthermore, the current fishing situation under the assumption of ontogenetic habitat shifts of resources may result in the natural protection of older more fecund fish (in areas outside the main fishing area) and hence may decrease the vulnerability to overfishing (Hixon et al., 2014). However, another explanation for the low biomass of large and medium predatory fish and the small mean length of the catch could be a long history of high fishing pressure in the bay. This is said to be the general case in other East African coastal areas such as in different parts of Kenya (McClanahan et al., 2008b). Results demonstrate that the use of dragnets in Chwaka Bay indeed has a strong negative impact on large *other carnivorous fish* and pelagic fish, and this high fishing pressure could thus have led to an overall reduction in biomass of these groups. However, when comparing the mean trophic level of the catch, the catch diversity and the mean length of the catch of the different fishing gears the Chwaka Bay fishery seems less deteriorated then the Kenyan coastal fishery. In particular, the catch of spear fisher in Chwaka is mainly comprised of higher trophic level species such as octopus and other carnivorous fish, while in Kenya and also Madagascar this gear is largely targeting herbivorous fish in the system, which are often less valuable (Davies et al., 2009; McClanahan and Mangi, 2004), indicating a major difference in the state of the fishery. The CPUE of dragnets, traps, handlines and spears is higher in Chwaka than in Kenya and Madagascar (Davies et al., 2009; Samoilys et al., 2017; Tuda et al., 2016). Since the coastal setting and the type of fishery is very similar in these countries, the higher catch per unit of effort in Chwaka bay could be indicative for a healthier, more productive system.

Results of the MTI routine demonstrate that traps and dragnets have strong negative impacts on the Chwaka Bay food-web and in particular on the key species. While the overall impact of dragnets on the system is stronger, it is also more evenly distributed among the different functional groups. In contrast, traps target to a large extend herbivores in the system, which has also been observed in other East African fisheries (Cinner et al., 2009; Mbaru and McClanahan, 2013). However, very little attention has been given to assess the implications of heavy fishing pressure from traps. Strong removal of herbivores can have detrimental effects on the structure of reefs and seagrass meadows (Hughes et al., 2007; Moksnes et al., 2008), as the reduction in macroalgae consumption by herbivores, allows the overgrowth of corals and seagrasses (Gullström et al., 2008). MTI results show that the impacts of herbivorous fish on the biomass of macroalgae is extremely low, indicating a loss of top-down control. Furthermore, traps are also the major cause of the reported overexploitation of the key species *S. sutor* and *L. lentjan* (Rehren et al., submitted*a*). This contrasts with the believe of many fishermen that dragnet fisher are the major cause of the reduction in fish biomass including key species (Pers. comm.). The only key species which experiences the strongest negative impacts from dragnets is *L. fulviflamma*. However, this species does not show any signs of overexploitation and is caught around its optimum length at first capture (Rehren et al., submitted*a*). The strong fishing impact of traps on populations of *S. sutor* and *L. lentjan* has also been shown for the Kenyan fishery (Hicks and McClanahan, 2012), but has not caught much attention yet. Dragnets are also reported as the major threat for growth overfishing in the bay by targeting immature fish (de la Torre-Castro and Rönnbäck, 2004; de la Torre-Castro et al., 2014), a situation that is similar to the Kenyan coastal fishery (Mangi and Roberts, 2006). However, traps have a highly similar mean length of the catch and handlines catch even smaller specimens.

Spears and handlines have little effects on the food-web, except for their main target group *octopus* and *squids*, which are heavily impacted. Consequently, these fisheries require much more attention and further stock assessments are needed.

4.4.3. Management challenges of the multigear fishery of Chwaka Bay

The habitat and system configuration of Chwaka Bay makes it an important and productive fishing ground providing stable year-round fishing yields. The bay seems near its exploitation limit, since the fish biomass is already highly reduced and certain fish groups (e.g. herbivores, pelagic fish) show great decreases in abundance. This is in accordance with the current high exploitation rates of the key species (Rehren et al., submitted*a*). However, when compared to the Kenyan coastal fishery, which shows strong indications of an unsustainable use of its resources (McClanahan et al., 2008b; Samoilys et al., 2017), the Chwaka Bay fishery seems to be less mature. The use of gears and the choice of mesh sizes seems to reflect the benthic-driven and nursery character of the bay, with traps and dragnets being the gears of highest impact on the food-web. Both gears potentially destabilize the ecosystem by reducing top-down control and herbivory on macroalgae. Furthermore, traps have the strongest impact on the key species of the bay, which contribute largely to the annual yield and therewith to the overall earnings. This strong fishing impact has already led to an overexploitation of *L. lentjan* and *S. sutor* (Rehren et al., submitted*a*). Although the strong impacts of traps on key species and herbivores seems to also hold for the Kenyan coastal fishery, the general focus of management recommendations lies on the ban of dragnets and spear guns (Hicks and McClanahan, 2012; Mangi and Roberts, 2006; McClanahan and Mangi, 2004). However, we argue that fishing effort of traps in Chwaka Bay is too high and that a reduction of trap fisher could maintain the harvest of these valuable key species, ensuring a large part of fishermen's income. Furthermore, we think that the total ban of dragnets as the most important measure for the fisheries in East Afirca`s coastal waters (Mangi and Roberts, 2006; McClanahan and Mangi, 2004) is not a viable management measure for the following reasons: 1) too many fishermen depend on their use for their livelihoods; 2) a redistribution of dragnet fisher to other gears would greatly increase the already very high number of boats in the bay, possibly causing further spatial use conflicts in the bay; 3) a redistribution of the dragnet fisher to other gears will have unforeseen effects on the food-web, which would need to be evaluated prior to the implementation of this measure. However, while the total ban of dragnets does not seem feasible for the above reasons, their use should be minimized due to their destructive effects on the environment. Thus, a partial redistribution of dragnets to longlines and handlines seems to be a reasonable management scenario, since these two gears are more profitable and the number of fishermen using these gears is still quite small. Spear guns have been identified as being highly destructive for the environment (Mangi and Roberts, 2006). Furthermore, in Kenya this gear seems to be highly impacting herbivores, similar to traps, and has been recommended for management restrictions (Cinner et al., 2009). In contrast, in Chwaka bay, this gear seems to be more selective, with little overlap with other gears and low impacts on the food-web. However, as their impact on the functional group *octopus* is critically high, we recommend an evaluation of the sustainability of the octopus fishery. Since the resource biomass of fish in the bay is already low and some of the key species show signs of overexploitation, the Chwaka Bay food-web does not provide scope for further expansion of the fishery. Thus, the entrance of new people into the fishery should be restricted, the use of dragnets minimized, partially by redistributing them to longlines and handlines and the use of traps in the bay should be reduced to avoid the collapse of key species.

CHAPTER V - Simulating Management Scenarios

Chapter V.

Ecosystemic and economic effects of different fisheries management scenarios in Chwaka Bay (Zanzibar): a modelling study using EwE

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Abstract

The increasing use of destructive pull seine nets (e.g. beach seines) throughout the Western Indian Ocean region has received great attention and management efforts, given its environmentally harmful as well as unselective character. Despite gear exchange programs and enforcement initiatives in Chwaka Bay (Zanzibar) this type of gear locally called dragnets, is steadily increasing and currently represents a highly important job provider for the local community. Within this context, we simulated the relative changes in biomass structure of the different functional groups and the relative changes in catch and profit of the different gears using a recently constructed *Ecopath* model of Chwaka Bay under four scenarios: 1) a complete ban of the dragnet fishery, 2) a reallocation of dragnet fishermen to other gears, 3) the overall increase in fishing effort, if no management measures are enforced and 4) a combination of effort reduction and reallocation of dragnets. Simulations suggest that the most beneficial scenario for the ecosystem and profits of fishermen would be the ban of dragnets without reallocation. However, it leaves 58 % of overall fishermen without jobs. In contrast, a complete reallocation of dragnet fishermen proportionally to the other gears would lead to decreases in overall fish biomass and individual profits by 38 % and 52 %. More importantly, some of the main target species approach zero by the end of the simulation time. In contrast the complete reallocation of dragnet fishermen to other gears would lead to strong reductions in biomass of target species as well as individual profits of fishermen. Simulations suggest that only 36 % of dragnet fisher can be reallocated if individual profits and the biomass structure of the ecosystem is not to fall beyond critical levels (-20 % and -30 % respectively).

Keywords: *Ecosim*; Chwaka Bay; artisanal fisheries; fishing effort reallocation; dragnets

5.1. Introduction

Artisanal fisheries are one of the mayor livelihood and protein suppliers in many coastal areas of East Africa (Wamsley et al., 2006; McClanahan et al., 2013). The large number of people directly depending on coastal fisheries highlights the importance for a sustainable management of its resources (Cochrane and Garcia, 2009; FAO, 2015). Decreasing catch rates, high juvenile retention rates and the increasing use of destructive gears raises concerns for a sustainable use of fisheries resources in many East African coastal areas (de la Torre-Castro et al., 2014; McClanahan et al., 2008b; Nordlund et al., 2013). Particularly, the wide spread use of beach seines receives great attention and is the focus of management in a variety of different coastal zones (de la Torre-Castro and Lindström, 2010; McClanahan et al., 2008b; RGZ, 2010; Wallner-Hahn, 2016; McClanahan and Mangi, 2001; Cinner, 2010). Despite the fact that this type of net is officially declared illegal in many areas, its use shows an upward trend (de la Torre-Castro et al., 2014; Kincaid et al., 2014). Beach seines have shown to be highly destructive, since nets are dragged over the sea floor destroying important habitats such as corals and seagrasses (Jiddawi and Ohman, 2002; Mangi and Roberts, 2006). Furthermore, beach seines are among the least selective gears (McClanahan and Mangi, 2004; Davies et al., 2009; Rehren et al., submitted*b*) and fisher often use small-mesh sizes leading to high juvenile retention rates (de la Torre-Castro et al., 2014; Hicks and McClanahan, 2012).

The combat of beach seines has been successful in different parts of East Africa. For instance, in Kenya the use of beach seines in the southern coast was effectively banned between 2002 and 2004 (McClanahan et al., 2008b). Within the Tanga Coastal Zone Conservation and Development Programme in 1994 a combination of the removal of illegal gears and a gear exchange program, facilitated the reduction of beach seines and other destructive fishing practices in the Tanga coastal zone (Wells et al., 2010). Gear exchange programs have also been partly successful in Mafia Island, where through the implementation of the Mafia Island Marine Park the use of beach seines was completely eradicated, while other pull nets are still in use (Kincaid et al., 2014).

However, in other areas the prohibition of these destructive seines nets did not proof to be successful. Such an example is the case of Chwaka Bay, where the use of dragnets has been prohibited officially since the implementation of the Zanzibar Fisheries Act in 1988 and the specific Chwaka Bay by-law in 2001. Dragnet boats differ from other gears such as handlines and traps in that they represent a small "fishing company". While trap and handline fisher generally own their gear and fish with on average 2 people per boat, a dragnet boat operates with on average 9 fisher (Rehren et al., submitted*b*; de la Torre-Castro and Rönnbäck, 2004). Fisher are needed to cast the net, dive down to set it and drive fish into it. Hence, fishermen are practically hired by the boat owner and thus they don't need any financial means or fishing experience (de la Torre-Castro and Lindström, 2010). In a situation where there is a general lack of alternative livelihoods, excess to higher education and financial support, such fishing method represents an important job provider to the community.

All attempts that have been made to ban the nets from the bay, turned out to be fruitless. A gear exchange program was also conducted in 2005 within the framework of MACEMP[7]. The major reason for its failure was the insufficient number of gears that were provided for the large amount of fishermen using illegal gears (Gustavsson et al., 2014). Hence, the situation did not change, but the effort of dragnets is steadily increasing (de la Torre-Castro and Rönnbäck, 2004; Pers. comm.). Such failures can weaken the trust and cooperation that is necessary for a successful fisheries management (Batista et al., 2014). Furthermore, the high dependency of the Chwaka Bay's community on its fisheries resources, bears the risk that effort reductions might increase food insecurity. Logically, the management of Chwaka Bay's fishery needs to put individual profits of fisher and overall employment at the centre of decisions. More importantly, gear exchange programs could have unforeseen consequences for the ecosystem, due to a sudden shift in effort regime. For instance, the direct increase in fishing mortality of vulnerable target species or cascading effects such as the increase in predation mortality on lower functional groups of the system caused by reduced fishing mortality on predators (Bacalso et al., 2016).

Within this context, our study aims at simulating the effects of different management scenarios on the Chwaka Bay ecosystem and its fishing community. More specifically, we simulate the relative changes in biomass structure of the different functional groups and the relative changes in catch and profit of the different gears using a recently constructed *Ecopath* model of Chwaka Bay. We simulated four scenarios: 1) a complete ban of the dragnet fishery, 2) a reallocation of dragnet fishermen to other gears, 3) the overall increase in fishing effort, if no management measures are enforced and 4) a combination of effort reduction and reallocation of dragnets.

[7] Partnership initiated in 2006 between the United Republic of Tanzania, the Global Environmental Fund and the World Bank.

5.2. Material and Methods

5.2.1. The Chwaka Bay ecosystem

Chwaka Bay is a tropical bay system on the east coast of Unguja Island, Zanzibar (6°02’6°13’S, 39°24’39°36’E). The bay is shallow (3 m in the bay proper to about 20 m around the reef at the offshore border) and comprised of a mosaic of seagrass beds, mangroves and corals, and is strongly dominated by a large biomass of primary producers and invertebrate consumer groups. Due to its high productivity and diverse habitats, the bay gives rise to an intense multigear, multispecies nearshore fishery and the density of fishermen in the bay is with 7 fisher km^{-2} comparatively high. The local community highly depends on the fisheries resources for income and protein supply. The main fishing gears are basket traps, dragnets, handlines, spears and, to a minor extend, floatnets, longlines, fence and gillnets (for further details refer to Rehren et al., submitted*b*).

5.2.2. Use of *Ecosim* as modelling tool

Ecosim is a time-dynamic part of the software *Ecopath with Ecosim*, which allows simulating the effects of changes in fishing and environmental conditions on the biomass dynamics of functional groups over time (Christensen et al., 2008). The equations are derived from the *Ecopath* master equation:

$$\frac{dB_i}{dt} = g_i \sum_j Q_{ji} - \sum_j Q_{ij} + I_i - (M_i + F_i + e_i)B_i \qquad (1)$$

where dB_i is the rate of change in biomass over time, g_i is the net growth efficiency(production/consumption ratio), I_i is the immigration , M is the natural mortality rate and e_i is the emigration. Qij are the feeding rates of each predator j on its prey i and are calculated based on the foraging arena theory (Ahrens et al., 2012), where B_i is separated in vulnerable and invulnerable compartments:

$$Q_{ij} = \frac{v_{ij} a_{ij} B_i B_j T_i T_j S_{ij} M_{ij} / D_j}{v_{ij} + v_{ij} T_i M_{ij} + a_{ij} M_{ij} B_j S_{ij} T_j / D_j} \qquad (2)$$

where a_{ij} is the effective search rate, T_i and T_j are the relative feeding times of prey i and predator j, S_{ij} are the seasonal or long term forcing effects, M_{ij} are the mediation forcing effects and D_j is the effect of handling time as a limit to consumption rate. The transfer rates vij are the most important parameters and represent the rate at which prey i shifts from a vulnerable state to an invulnerable state. These parameters determine whether the system is top-down, intermediate or bottom-up controlled. Hence, the model is very sensitive to these vulnerability rate settings and they are usually calibrated by fitting the model to time series data of one or more functional groups (For more detailed information, see Christensen et al., 2008). However, the availability of time series data for tropical fisheries systems is often lacking and for such situations it has been proposed by Cheung (2001) and Cheung et al. (2002) to set the vulnerabilities (v_i) of a prey or predator proportional to its trophic level (TL):

$$v_i = 0.1515TL_i + 0.0485 \qquad (3)$$

In the absence of sufficiently long time series data for Chwaka Bay, this linear relationship was used to set the vulnerabilities of the different functional groups.

5.2.3. Use and conservation scenarios

A recently constructed *Ecopath* model of the Chwaka Bay ecosystem (Rehren et al., submitted*b*) was used as reference to simulate four different fishing effort scenarios (see below). The Chwaka bay model is comprised of 28 functional groups ranging from primary producers to pelagic fish. The model is mainly rooted in local data and has a pedigree of 0.53.

Scenario I assumes that the current trend in effort will continue unregulated with no further management measures in place. Data to estimate the current trend was obtained from the Department of Marine Fisheries Resources. Since detailed data on the total amount of boats per year active in Chwaka Bay is not available, we inferred the trends in effort based on the total number of boats landed per month for all years that were accessible, this included: July – December 2009 (Missing September), January – May 2011, January – December 2012 and January – June 2013; we then extrapolated the increase in boats using a logarithmic trend line until the year 2025 (further information see S.5.1.). To estimate the increase in the number of boats for each particular gear, we assumed the ratio between the different gears to remain constant over time. We did not predict the increase in effort for fence fisher, because their use is restricted to the intertidal area and taking into account the intense cultivation of seaweed in the bay's intertidal zone it is not clear to what extend an increase in this fishery would be spatially limited.

In *Scenario II* we simulated a stepwise elimination of the illegal dragnet fishery in the bay. The starting value of the simulation was the number of dragnet fishermen operating in 2014 and this value was reduced each year by 20 %. In *Scenario III* we simulated the proposed redistribution of the dragnet fishery to other legal gears. For this we reallocated all dragnet fishermen that were active in 2014 to the other gears in proportion to the relative effort of those gears. Fence were not included in the redistribution due to the reasons mentioned above. Furthermore, we did not redistribute dragnet fisher to spears, because the fishery data did not allow for a differentiation between the legal wooden sticks and the illegal spear guns (Fisheries Act Zanzibar, 7 2010, RGZ, 2010), for the reasons mentioned above.

Furthermore, we simulated an alternative management scenario (*Scenario IV*) in which we aimed at reallocating as many fishermen as possible without a) causing negative biomass changes of more than 30 % for any functional group and b) causing decreases in individual profits over 20 %. Both thresholds are set arbitrarily, but we find them to be reasonable, because a) a 30 % change in different biological indicators for overfishing have been repeatedly proposed to be used as warning thresholds (Hall and Manprize, 2004; Link et al., 2005); and b) many fishermen are already earning close to the poverty line (de la Torre-Castro and Rönnbäck, 2004) suggesting that they cannot withstand strong reductions in their profits.

The simulation time of the different scenarios was set to 50 years for all scenarios except for *Scenario I* where we only looked at the increase in effort over a period of ten years. All scenarios started after the first simulation year. Table 1 gives an overview of the relative change in fishing effort of the different gears used for the different scenarios.

Table 5.1. The change in relative fishing effort of all gears for Scenario I – IV (numbers in table are effort multipliers for each scenario).

	Scenario I	**Scenario II**	**Scenario III**	**Scenario IV**
	Past and future trends in effort	Elimination of dragnets without reallocation	Elimination of dragnets with reallocation	Alternative management approach
Dragnets	1.4	0	0	0
Trap	1.4	1	2.8	1.2
Handline	1.4	1	2.9	1.4
Spear	1.4	1	1	1
Longline	1.4	1	2.7	4
Gillnet	1.4	1	2.7	4.2
Floatnet	1.4	1	2.9	3.4
Fence	1	1	1	1

5.2.4. Changes in profit, catch, biomass and ecosystem structure

To compare the impacts of the different fishing patterns on the local community and the ecosystem, we compared the relative changes in biomass (t km^{-2} yr^{-1}), catch (t km^{-2} yr^{-1}), net profit (TZS km^{-2} yr^{-1}, 1 USD = 1663.2 TZS, 2014) and individual profit (TZS $fisher^{-1}$ $trip^{-1}$) of the different scenarios. Results for the comparison of the scenarios were taken from the simulation year in which values had stabilized (biomass) or in the case of *Scenario IV* from the last simulation year.

5.3. Results

Table 5.2. Percentage of fishing jobs lost, CPUA and changes in overall fish biomass as well as target groups biomass by the end of simulation time of Scenario I-IV.

	Jobs lost [% fisher]	CPUA [t km^{-2} yr^{-1}]	Overall fish biomass change [%]	Changes in total biomass of target groups [%]
Baseline	0.00	4.77	-	-
Scenario I	-	4.82	-17.00	-2.50
Scenario II	58.30	3.87	19.00	3.80
Scenario III	0.00	4.05	-38.00	-3.50
Scenario IV	36.80	4.63	10.00	1.10

5.3.1. Relative changes in biomass of target groups

The relative change in the biomass of selected target groups for *Scenario I-IV* is shown in Fig. 5.1. and in Table 5.2. the changes in overall fish biomass of the system as well as the changes of total biomass of target groups is listed. If, the current trend continuous (*Scenario I*), the biomass of the target groups decreases by 2.5 % with a reduction in overall fish biomass of 17 %. A reallocation of dragnet fishermen to other gears (*Scenario III*) results in an even stronger decrease in target group biomass and overall fish biomass (-3.5 %, -38 %, respectively). In contrast, the removal of dragnet fishermen (*Scenario II*) and the alternative management scenario (*Scenario IV*) leads to an overall increase in target groups by 3.8 and 1.1 %, respectively. And in both scenarios the overall fish biomass shows a relatively strong increase of 10 - 19 %.

The functional groups that are impacted the most by the reallocation of dragnets to other gears (*Scenario III*) are squids, *S. ghobban*, *L. lentjan* and *L. borbonicus*, since biomasses of these groups approach zero by the end of the simulation time. Other groups that show a strong decrease in biomass are *other herbivorous fish*, *other carnivorous fish*, *S. sutor* and *L. vaigiensis* (-32 % to -72 %). *L. fulviflamma, omnivorous fish* and *zooplanktivorous fish* are groups that benefit a redistribution of dragnet fisher by an increase of 51 - 84 % in biomass. The latter two functional groups also benefit under the fishing pattern of *Scenario I*, since biomass of these groups increase of 53.2 and 56.2 %. However, most of the other groups show negative changes, with very strong decreases in biomass of the functional groups *L. vaigiensis*, *L. borbonicus*, *other carnivorous fish*, *pelagic fish*, *other herbivorous fish*, *octopus* and *squids* (-32.5 % to -51.5 %). In contrast an elimination of dragnet fishermen without reallocation (*Scenario II*) results in a strong decrease of omnivorous and zooplanktivorous fish (-33 to -42 %) together with a small decrease in the biomass of *S. ghobban* (-10 %). In this scenario all other groups show a positive biomass change ranging from 3 % for *S. sutor* to 54 % for *pelagic fish*. In *Scenario IV* biomass increases are much stronger for most of the groups. For instance, the key species *L. fulviflamma*, *L. lentjan*, *L. vaigiensis*, *S. sutor*, *zooplanktivorous fish* and *other herbivorous fish* show an increase in biomass between 37 % and 60 %. In this scenario the biomass of *other carnivorous fish*, *pelagic fish*, *squids* and *squids* show relatively strong declines. However, these declines in biomass do not exceed 30 % for neither of these groups.

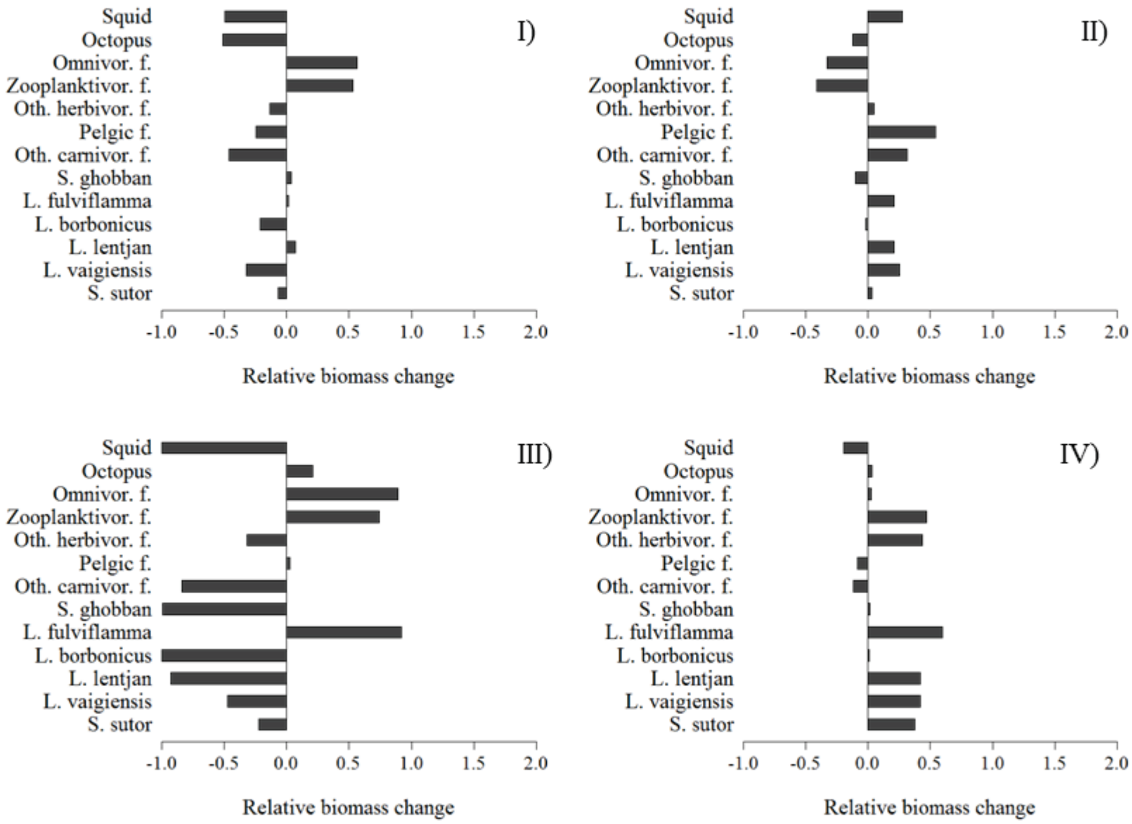

Fig. 5.1. Changes in relative biomass of selected target groups under scenario I-IV (simulation period is 50 years for scenarios II-IV, and 10 years for Scenario I).

5.3.2. Relative changes in catch and net profit of the different gears

Changes in catch and net profit of gears under *Scenario I-V* are shown in Fig. 5.2. Since the effort of dragnets is to a varying degree reallocated or removed, catch and profit of this gear is strongly reduced in *Scenario II-IV*. Furthermore, for these three scenarios the total catch decreases, with *Scenario IV* showing the lowest decrease and *Scenario III* the highest (Table 5.2). Under *Scenario III* floatnets benefit the most from the reallocation of dragnet fisher, since both catch and net profit increase by the factor 1.8. Similarly, the catch and net profit of longline and trap fishermen show a relatively strong increase of 17 % to 72 %, while the rest of the gears experience very little or negative changes (-105 % to -1 %). When removing all dragnet fisher (*Scenario II*), catch and net profit of the other gears increases between 6 and 100 %, with fence, gillnets and floatnet fisher benefitting the most. The latter two together with handline and longline fisher benefit even more under *Scenario IV*. Floatnet, gillnet and longline fisher for instance show an increase in catch and net profit of 213 - 274 %. The only gear that shows a loss in catch and net profit under *Scenario IV* are fence (< 10 %).

If the current trend continuous (*Scenario I*), only the net profit for trap fishermen increases (10 %). Although the catch of dragnets and floatnets are likewise showing a small increase (5.7 – 11 %), their profit decreases by 4.8 % and 26.3 %. The rest of the gears show a decrease in catch and net profit by 0.5 - 44.4 %.

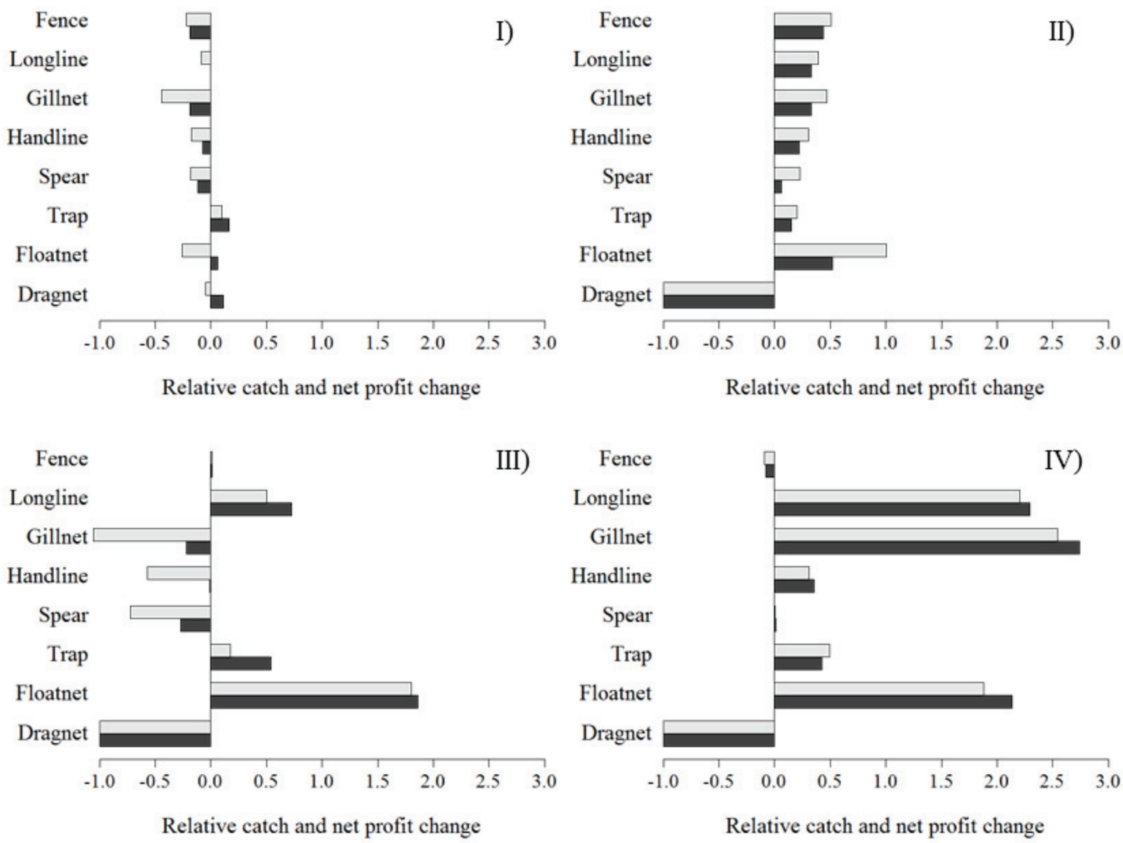

Fig. 5.2. Changes in catches (dark-coloured bars) and net profit (light-coloured bars) of the different gears under scenario I-IV (simulation period is 50 years for scenarios II-IV, and 10 years for Scenario I).

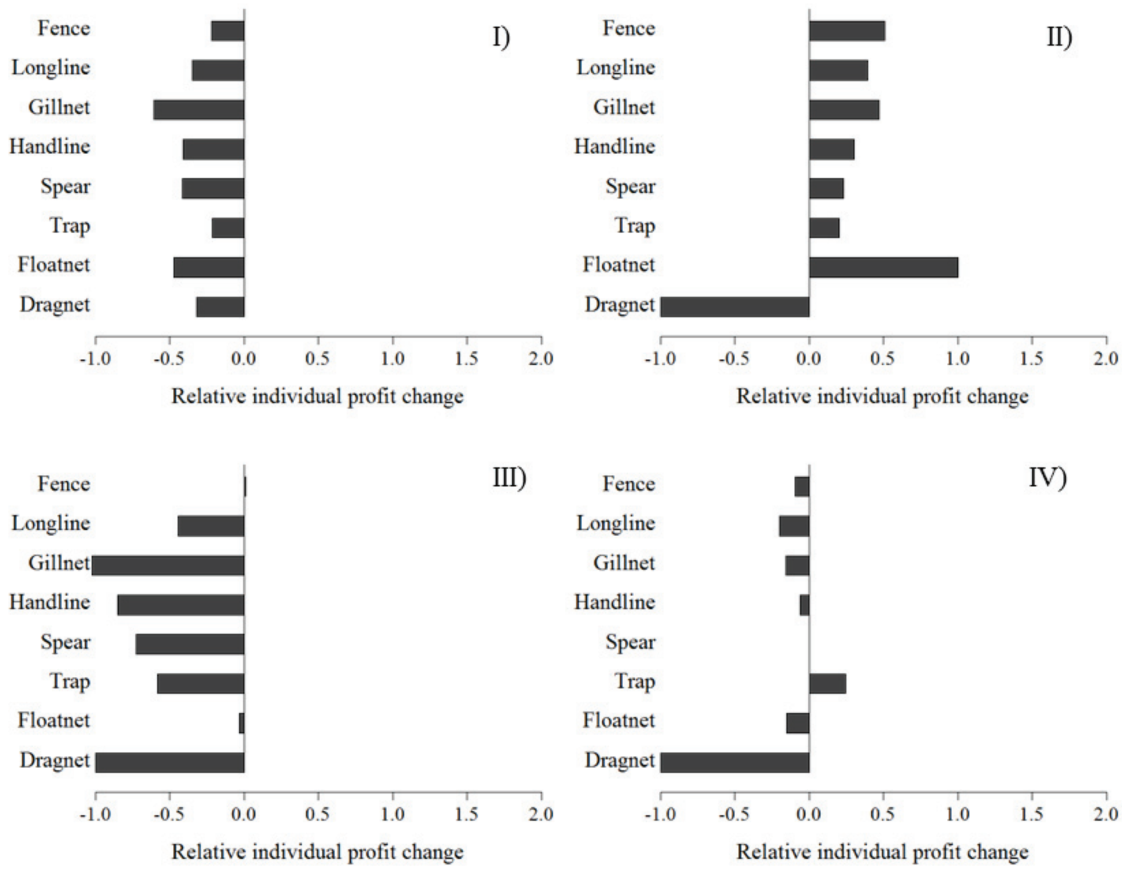

Fig. 5.3. Changes in individual profits of fishermen of the different gears under scenario I-IV.

5.3.3. Changes in employment and individual profit

The fishing community experiences the highest level of unemployment under *Scenario II* (Table 5.2), since all dragnet fisher are removed which equals to 58.3 % of the total number of fishermen operating in Chwaka Bay. However, the elimination of dragnets results in a relatively strong increase in individual profits of all other gears (20 to 100 %, Fig. 5.3.). In *Scenario III* the total number of jobs does not change, as all fishermen are redistributed to other gears. But none of the other gears benefits from the resulting increase in their effort, except the fence fisher (1.2 %). For trap, handline, spear and gillnet fisher the resulting loss in individual profit is very high (> 50 %). In *Scenario IV* we redistributed as many fisher as possible to the other gears without exceeding a biomass loss of 30 % for any functional group and exceeding an individual profit loss of 20 %. With these thresholds not all fisher could be redistributed and ultimately 37 % of the total number of fishermen operating in Chwaka Bay lose their jobs. Only trap fisher benefit from this scenario with increases in individual profits of 24 %.

5.3.4. Past and future trends of individual profits and biomasses

The relative changes in biomass and individual profits of *Scenario I* are depicted in Fig. 5.4. and Fig. 5.5.. The backward simulation resulted in biomass of target groups that are 2.3 to 30.3 % higher in 2009 compared to the baseline year 2014. Furthermore, the overall fish biomass is 9.9 % higher in 2009. After seven simulation years starting from the reference year 2014 (2021) the biomass of *octopus* and *squids* falls below 70 % of their original biomass and one year later the biomass of *other carnivorous fish* shows a similar decline of 29 %. In the same year (2022) the individual profit of the different gears dropped by 15 to 39 %, while the number of fishermen increased by 29 %, which is equal to 248 fishermen. Similar to the biomass, individual profits of all gears (except fence) are 10.7 to 38.6 % higher in 2009 than in the baseline year.

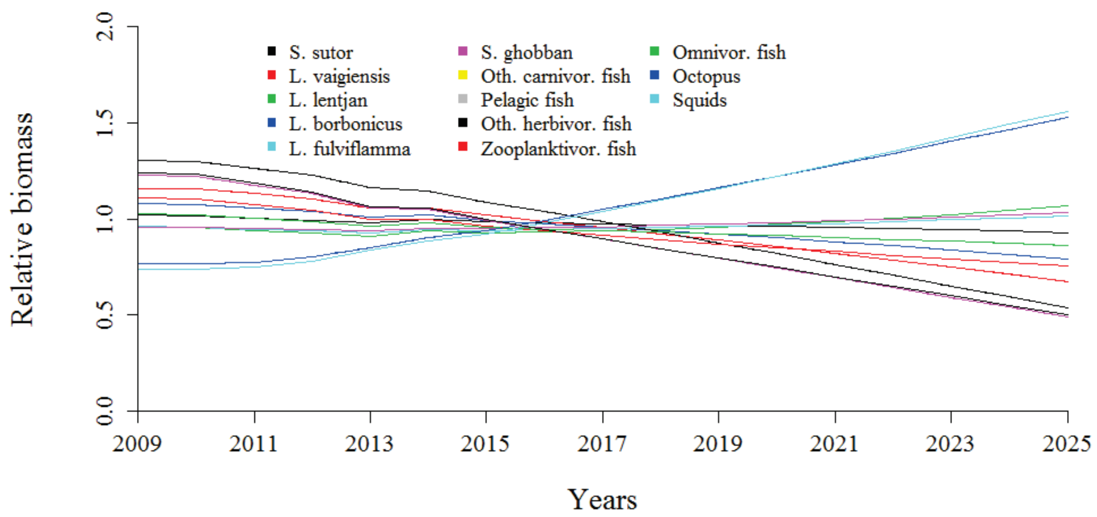

Fig. 5.4. Relative biomass changes over time using effort data from 2009, 2012 and 2013 and extrapolating the respective logarithmic trend to 2025.

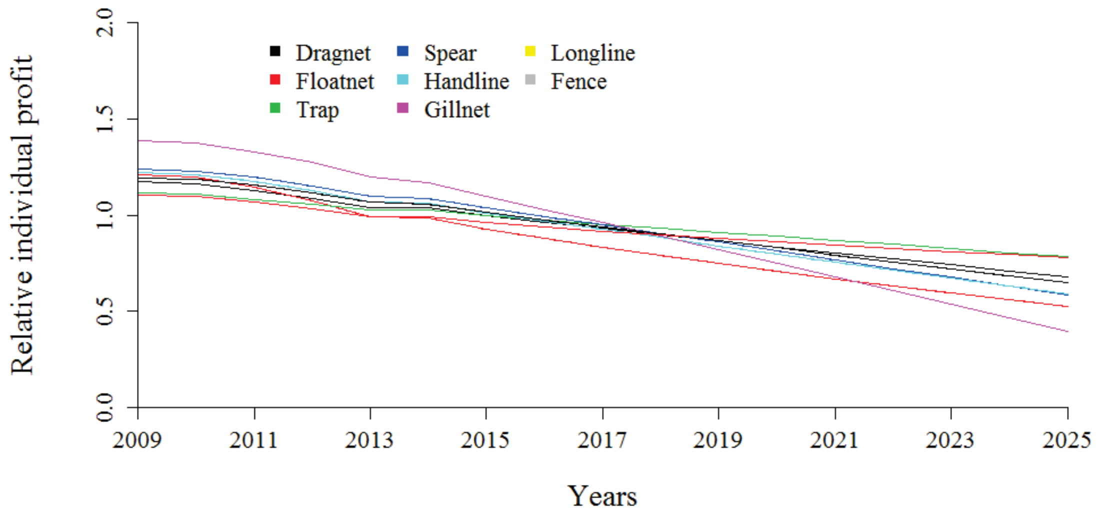

Fig. 5.5. Relative changes in individual profits of fishermen over time for the different gears using effort data from 2009, 2012 and 2013 and extrapolating the respective logarithmic trend to 2025.

5.4. Discussion

5.4.1. Past and future state of the fishery

The local community of Chwaka Bay is highly depending on its fisheries resources, since it is the main protein supplier and half of the inhabitants are directly involved in the fishery (Jiddawi, 2012). Fishermen report declines in catch rates and their subsequent income (Jiddawi, 2012, de la Torre-Castro and Rönnbäck, 2004) and due to its open-access character, the fishery experiences a steady increase in effort (Kathib and Jiddawi, 2010). This, together with findings from recent studies that some of the key species show signs of overexploitation (Rehren et al., submitted*a*), indicates that Chwaka Bay is experiencing an overcapitalization of its fishery.

Unfortunately, there is no long-term estimation of effort for any village of the bay, which makes it difficult to assess the catch per unit of effort over time. We chose a logarithmic regression line to represent the trend in effort using effort estimations from 2009 until 2013. However, in 2013 the effort dropped about 2 %, which could be indicative for a shift in the slope of the effort curve. However, the few data points together with the fact that population numbers of Zanzibar are increasing and that fishermen keep complaining about an increase in effort, led us to model the changes in fishing effort based on a logarithmic increase.

According to backward simulations a fisher in Chwaka Bay had on average a 16.6 % higher profit 5 years ago (2014, year of the base *Ecopath* model) , which is about 1300 TZS more (1 USD = 1645.3 TZS, April 2014) and is well above the extreme poverty line at that time (1.25 $, World Bank, 2008). In 2014, the profit of the different gears approaches the current extreme poverty line (1.90 $ day^{-1}, World Bank, 2015), with dragnets and floatnets earning only 3 and 23 cents more (1.90 $ day^{-1}, World Bank, 2015). In contrast to the better income conditions (and lower overall effort) of fishermen in 2009, the overall fish biomass and in particular the biomass of *pelagic fish* does not seem much higher (9-11 %). Fishermen in Chwaka Bay say that they used to catch much more pelagic fish than they do nowadays (pers. comm.), probably referring to a time long before 2009. Nevertheless, *other carnivorous fish*, *octopus* and *squids* were up to 30 % more abundant in the simulation year 2009, indicating that the increase in effort within the last 5 years could have had a strong impact on the biomass of these functional groups. Simulating increases in effort after 2014 suggests that the decline in fish biomass will likely intensify, leading to further losses in top-down control in the system. Within the next 5 to 10 years *octopus*, *squids* and *other carnivorous fish* could severely decline (-12.4 to -51.1 %). Despite the fact, that these 3 functional groups are affected most heavily by an increase in overall effort, spear and handline fisher do not show the highest decrease in profits. Floatnet fisher, which heavily rely on *pelagic fish*, experience the highest losses. Trap is the least affected gear, since it targets a wide variety of different species and only two of their target functional groups show biomass declines beyond 10 %. Hence it seems as traps are more resilient to the simulated increases in fishing effort.

Simulations also predict that, after 6 years, individual profits of the different gears will show a decline of 10 % or larger. This is in particular impacting the low income of floatnet and dragnet fisher, which will fall below 1 \$ day^{-1} by the end of the simulation in 2025. Furthermore, in 2025 fence, gillnet and spear fisher will earn below or very close to the current poverty line. A survey by Tietze et al. (2002) observed that declining catches and a decrease in income of coastal fisher in six tropical countries was accompanied with a stagnation or decline in the number of active coastal fisher. Furthermore, a study by Cinner et al., (2008) showed that almost half of all interviewed fisher would exit the fishery if catch rates decline by 50 %. This situation is likely to occur in Chwaka Bay, since the strong loss in individual profits (20 to 60 %) might push fishermen to turn fully or partly to other livelihoods. In fact, during the data collection in 2014, two of the assisting fishermen started to partly engage in farming due to the lack of sufficient income. Although it should be noted, that our estimates do not account for inflation and changes in market price of the different species, simulations clearly indicate that a continued increase in effort, has severe implications for the catch and profits of Chwaka Bay's fishermen to a point where the income from fishing may not any longer feed fishing households. It has been shown elsewhere, that the sustainability of fisheries stocks together with profits for fishermen decreases strongly once fishing effort increases significantly (Purcell and Pomeroy, 2015). With the trend in effort used in our scenario, this situation will largely intensify. The absence of actions taken by the government to control fisheries access, together with the increasing demand for fish will likely exacerbate the dilemma of overcapitalization. A technological development of the fishery (e.g. large boats, more engines, greater horse power, enhanced gears) could further reinforce the problem, leading to much higher pressure on Chwaka Bay's fisheries resources (Pomeroy, 2012), unless the fishing area is extended further offshore.

5.4.2. Management option

The elimination of destructive gears, in particular dragnets, is often one of the central issues in the management of fisheries resources in East Africa (de la Torre-Castro and Lindström, 2010; Mangi and Roberts, 2006; McClanahan et al., 2008b; Tobey and Torell, 2006). In Chwaka Bay, despite several attempts to ban this gear, its use is steadily increasing (de la Torre-Castro and Rönnbäck, 2004). Our results show that a successful ban of dragnets would have strong positive impacts on the ecosystem.

Although the overall fish biomass remains low, it shows an increase of 18.8 %. In particular the biomasses of *L. lentjan*, which shows signs of overexploitation (Rehren et al., submitted*a*), and the ecologically important *pelagic fish* and *other carnivorous fish* benefit from an elimination of dragnets. However, the release of fishing pressure on the latter two functional groups sets off trophic cascades and leads to the reduction in the biomass of prey groups. The predation mortality of *octopus*, for instance, is highly increased, while this group does not experience any release in fishing mortality and as a consequence suffers from strong biomass reductions. In addition to the benefit for the ecosystem, the individual profit of the fishermen increases strongly by 20 % to 40 %. This result is crucial, regarding the fact that many fishermen are concerned about decreases in their daily income (Jiddawi, 2012; de la Torre-Castro and Rönnbäck, 2004). The individual profit response to our simulation is in line with observations of actual fishing communities in Kenya, where an increase in profits and catch per unit of effort was observed after banning illegal seine nets (e.g. beach seines, McClanahan et al., 2008b; McClanahan, 2010). If taking into account the benefits to the ecosystem and the profits of fishermen, the elimination of dragnets clearly seems to be the most successful management option. However, such elimination reflects a strong effort reduction and leaves 58 % of fisher in the bay without jobs. It has been shown, that the implementation of management plans to reduce fishing effort in many tropical artisanal fisheries, is highly complex and very difficult, because of the high dependency on fisheries resources and the lack of alternative livelihoods (Daw et al., 2012; Salayo et al., 2008). In order to successfully ban all dragnets from the bay, the government would need to provide appropriate management plans such as buy-back schemes, and programs to generate alternative forms of income. Seaweed farming and tourism have been proposed but seem to represent insufficient solutions (Eklöf et al., 2012; Gustavsson et al., 2014). With little other livelihood opportunity, dragnet fishermen have no alternative but to continue to fish and if necessary to decrease their mesh size or increase their effort. Dragnet fishermen of Chwaka Bay perceive the ban of dragnets without the provision of other possibilities as unfair, since it leaves them no other choice than breaking the law (Wallner-Hahn et al., 2016). Furthermore, in Chwaka Bay many of the fishermen are directly depending on fish as a source of protein (Jiddawi, 2012), because meat is expensive. Even if income from fishing is insufficient to secure all needs of the household, fishermen usually still get enough fish for home consumption. Thus, management actions including the ban of dragnets might increase food insecurity and could have severe health implications for the local community.

Despite the failure of the gear exchange program conducted within the framework of MACEMP (Gustavsson et al., 2014), dragnet fishermen in Chwaka Bay say they are willing to change to other gears (Walner-Hahn et al., 2016). However, our simulation shows that a complete redistribution of dragnet fishermen could have severe consequences for the ecosystem. Since the effort of the main gears in use (traps and handlines) is already high, a reallocation of fishermen using these fishing gears will likely lead to the collapse of their key species such as *squids*, *L. borbonicus* and *S. ghobban*. The results of *Scenario IV* indicate that the key functional groups of traps and handlines only tolerate a small effort increase for these two gears (1.2 - 1.4 fold) if strong biomass and individual profit reductions (> -30 %, > -20 %, respectively) are to be avoided. It should, however, be noted that in our model we do not distinguish between different hook sizes or bait used. As a consequence those handlines that primarily target other species than squids and the small emperor *L. borbonicus*, could possibly allow for a greater increase in effort.

A recent study also suggests that the current effort under the current length at first capture is not sustainable for three of the key species; inter alia, *S. sutor*, one of the most important species in the region (Rehren et al., submitted*a*). A further study has shown that the strongest impact on this species is induced by the trap fishery (Rehren et al., submitted*b*). Outcomes from the gear exchange program conducted in the coastal fishing communities of Tanga within the framework of the Tanga Coastal Zone Conservation and Development Programme supports our findings. During the time of the program herbivorous fish declined, accompanied by a decrease in the use of destructive gears (e.g. beach seines) and an increase of trap effort (Wells et al., 2010). Not only are many dragnet fishermen of Chwaka Bay reluctant to change to trap fishing (Wallner-Hahn et al., 2016), our results also question the sustainability of a large increase in the use of traps in the bay. In addition to biomass reductions of certain key functional groups, a complete redistribution will likely result in a lower overall fish biomass in the bay and as a consequence in the loss of key stone species and reduced top-down control. The decline in biomass of target groups will result in declining catches for most of the gears. More importantly, simulations indicate severe losses in individual profits of trap, handline, spear and gillnet fishermen (up to -50 %). Thus, it seems that a gear exchange program aiming at a complete reallocation of dragnet fishermen is not possible without severely impacting the ecosystem and putting Chwaka Bay's resources at risk.

In contrast to the main gears, floatnets, gillnets and longlines, at present, have a very low fishing effort in the bay and our simulations indicate that a strong increase (3.4 - 4.2 fold) of these gears is possible without strong reductions in the biomass of functional groups (> 30 %) and individual profits (> 20 %). Hence, if dragnet fishermen are to change to legal fishing practices, the focus could be on these three fishing gears. This is supported by the fact that generally fisher on Zanzibar say they would like to fish with more advanced gears (e.g. longline and gillnets) targeting more valuable fish (e.g. large pelagics, Thyresson et al., 2013). It is not surprising that dragnets are not the first choice for a fisher, considering that the income of a dragnet fisher (except for the boat owner, as he gets a bigger share) is less than for other gears (Rehren et al., submitted*b*) and that dragnet fishing is physically more demanding. The reason that they do not use the preferred gear is a lack of access and financial means (Wallner-Hahn et al., 2016). However, there are several factors that have to be looked at, when considering the use of gillnets and longlines as the basis for a redistribution of dragnet fishermen. It has to be noted, that longlines and gillnets are gears that are only partially applied within the study area limits. Many days these fishermen go up North to Kiwengwa (approx. 24 km) or go offshore to fish. In our profit estimation we do not account for variation in fuel costs due to difference in distance between fishing grounds. Thus, further analysis is needed that accounts for differences in sailing related costs. Gillnets are also shown to have higher maintenance costs (Mangi et al., 2007). In addition, fishermen using gillnets or longlines report that they repeatedly come back empty-handed, while fishermen using one of the main gears (e.g. traps, spears, dragnets and handlines) say that they at least catch enough for home consumption or bait use. Lastly, many longline and gillnet fisher change gears during seasons with strong winds and unfavourable environmental conditions. All of these factors need to be taken into account when planning and implementing gear exchange programs. The financial support for an exchange of dragnets to gillnets or longlines might go beyond the provision of gear. Successful programs might need to include an initial capital for fuel costs, financial means to buy bigger boats and compensation of days with no catch. Furthermore, training of how to use these gears might be necessary.

Finally, our analysis questions the sustainability of a complete reallocation of dragnet fishermen using any of the other gears. The 30 % biomass reduction limit together with the 20 % individual profit limit, allowed for a 3.4 - 4.2 fold increase in gillnets, longlines and floatnets and a 1.2 - 1.4 fold increase in traps and handlines, but is not enough to reallocate all dragnet fishermen. Although, this is a considerable improvement over the complete ban of dragnets without reallocation, it still leaves 37 % of fishermen unemployed. The strong focus on the elimination of dragnets in the bay is mainly due to the dragging technique of this gear, which is said to strongly damage the habitat. However, a recent study by Gullström et al. (2008) found that the overall seagrass cover of Chwaka Bay has remained relatively stable between 1987 and 2003 despite the very frequent use of dragnets in these areas. Considering these finding, and the high job provision by dragnet boats and the lack of alternative livelihoods, we suggest the implementation of an effort control for this gear, by reallocating some of the fishermen to other gears and then prohibiting the operation of new dragnet boats in the bay. Given the fact that gear management, particularly the ban of destructive gears, is one of the most accepted management measures in East Africa (McClanahan et al., 2012, 2008a), the control of access into the dragnet fishery has a great probability of being effective. The impacts of the reallocation of dragnet fisher on the catch and profit of other fishermen should be monitored carefully, and if successful, measures for a further redistribution can be implemented. The effort control in the bay is particularly important, given the fact that our analysis indicates an already existing overcapitalization of Chwaka Bay's fishery. However, it has been shown that the reduction in fishing effort may only be short-lived (under an open-access regime like in Chwaka Bay), since subsequent increases in individual profits will attract new fishermen (Pomeroy, 2012). And that might be particularly relevant for a redistribution of dragnet fisher, since the reduction in dragnet effort will increase individual profits of the remaining fisher as our results indicate. Under a situation of poverty and the lack of alternative livelihoods, such management plans will require poverty alleviation strategies and generation and diversification of livelihoods (Batista et al., 2014; Kittinger et al., 2013).

CHAPTER VI - Synthesis and Conclusion

6. Synthesis and Conclusion

The central aims of this thesis were to evaluate the status of the Chwaka Bay fishery from a single-species as well as from an ecosystem perspective and to investigate different management alternatives for the sustainability of the bay's fishery. Thereby this thesis is aimed at providing a basis for the management of Zanzibar's inshore fisheries. In the following sections I summarize the principal findings of the thesis and discuss the limitations of the data and methodology used here in providing an understanding of the status of the fishery as well as in providing useful target limits for management. Further, I discuss potential management recommendations and future research directions. Finally, I provide recommendations for the improvement of national data collection on Zanzibar.

6.1. State of the inshore fisheries of Chwaka Bay and potential management options

6.1.1. General state of the fishery

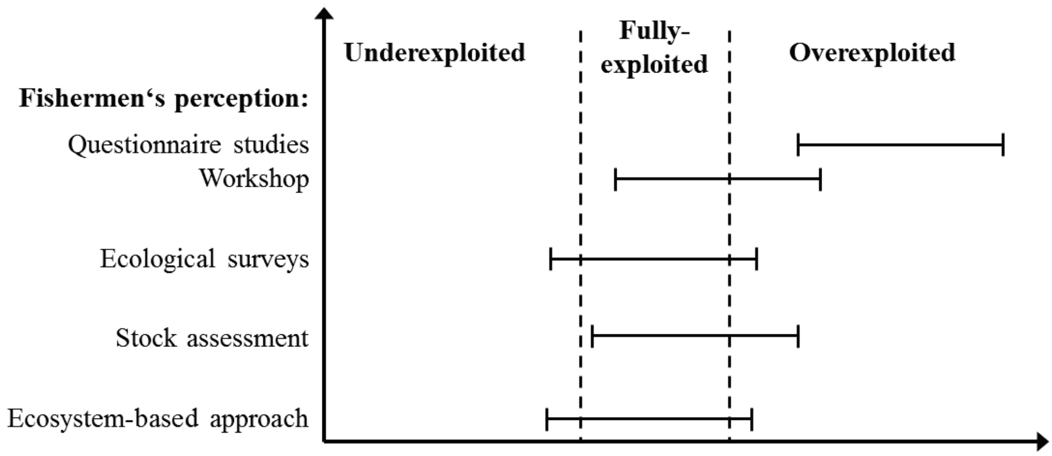

Fig. 6.1. Overview of the state of Chwaka Bay's inshore fishery according to fishermen's perception, ecological surveys (Aller et al., 2014; Fröcklin et al., 2014)*, stock assessment (Chapter III) and the ecosystem-based assessment (Chapter IV and V). Fishermen's perception were assessed through questionnaires with fishermen in 2002 and 2014 (de la Torre-Castro and Rönnbäck, 2004; Geere, 2014*) *and through a participatory workshop conducted with fishermen from Uroa, Marumbi and Chwaka village in September 2016.*

The approach used in this study helped to lift the fishery of Chwaka Bay out of a complete data-poor situation (Only anecdotal information) into a fishery that could be classified as low to medium studied. Thereby this study provides managers with useful information for setting management measures. The long field period, the data collection together with fishermen and the proximity to the fishing community through staying and living in the community, allowed me to gain a deeper insight into the problems and dynamics of the fishery. This represented a unique opportunity to grasp the key dilemmas of the fishery; I strongly believe that in order to develop research into a direction that ultimately serves management as well as fishermen, such intense field work and the proximity to the fishing community is imperative.

The concern of a general overexploitation of Chwaka Bay's resources could not be confirmed by this study. The Chwaka Bay ecosystem itself is relatively mature suggesting that the current level of fishing pressure has not yet pushed the system towards a highly disturbed, immature state. Furthermore, the fishery of Chwaka Bay seems less deteriorated than other heavily fished systems in the WIO region such as the Kenyan fishery. However, we do see a strong footprint of the fishery on the system. For instance, the high transfer efficiencies and the overall low fish biomass are indicative for a strong exploitation. Particularly, the low impact of sea urchin predators and herbivorous fish on their prey groups is worrying. In addition, for some of the fish groups recruitment overfishing is occurring, as results from the stock assessment show. Overall these findings suggest that the fishery of Chwaka Bay is fully exploited with some groups experiencing overfishing. The fishery's status thus leaves no scope for further expansion. An overview of the state of Chwaka Bay's fishery according to the findings from this study as well as outcomes from ecological surveys (Aller et al., 2014, Fröcklin et al., 2014) and questionnaires (de la Torre-Castro and Rönnbäck, 2004; Geere, 2014) conducted in the bay are depicted in Fig. 6.1.. The findings of this study are crucial, as they clearly show that a combination of declining catch rates with the use of small-mesh sizes and illegal gears does not necessarily reflect a state of overfishing. This situation has already been highlighted by other authors (see e.g. Kolding et al., 2014). However, due to the lack of knowledge on actual stock and ecosystem state throughout the WIO region, it is likely that, similar to the Zanzibar case, these insufficient indicators are frequently used to evaluate the state of a given fishery. Furthermore, in Chapter II we highlight that most of the research effort conducted to evaluate the impacts of fishing on ecosystems in Zanzibar fails at providing managers with appropriate information to set management plans or tangible reference points. This is essential given that the institutional and financial capacity of fishing authorities in many WIO countries is strongly limited and thus fisheries management could benefit largely from well-designed academic research studies. I recommend that fisheries-related research in the WIO region needs to be designed in a way that a) ecological indicators are linked with input measures (e.g. fishing effort) and b) they provide managers with sustainable reference points.

In contrast, to the situation in Chwaka Bay, findings of Chapter II suggest that several of the reefs located at the West Coast (i.e. Chapwani, Changuu, Pange, Bawe) already experience a fishing pressure that has pushed the biomass of many target fish beyond sustainable limits. In Table 6.1. the total number of vessels as well as the number of vessels per landing site for each district are depicted (Kathib and Jiddawi, 2010). It shows that the fishing pressure on the West Coast (Mjini and Magharini) is relatively high. However the two districts Micheweni and Mkoani on Pemba experience an even higher fishing pressure indicating that the coastal resources within these sites might experience a similar state of overexploitation. In contrast, the remaining four districts (that exclude Chake Chake on Pemba) likely experience a similar or a slightly higher exploitation level when compared to Chwaka Bay. While this is only a crude comparative examination, it points towards the need to examine the sustainability of the fisheries of Micheweni and Mkoani. Since fishermen are reporting a decrease in catches for a long time, a further increase in fishing effort will likely exacerbate the dilemma throughout Zanzibar. Management actions are thus needed to 1) reduce the pressure on those groups that are currently enduring excessive fishing mortalities and 2) improve the livelihood of the fishing community.

Table 6.1. Number of vessels per district and landing sites reported in 2010 for Unguja and Pemba (Kathib and Jiddawi, 2010).

Island	District	Number of Landing sites	Total number of vessels	Number of vessels per landing site
Pemba	Micheweni	11	1186	107.8
Pemba	Mkoani	16	1376	86.0
Unguja	Magharini	15	1254	83.6
Unguja	Mjini	6	392	65.3
Unguja	Kaskazini A	31	1274	41.1
Unguja	Kaskazini B	15	550	36.7
Pemba	Wete	26	792	30.5
Unguja	Kusini	22	645	29.3
Unguja	Kati	34	724	21.3
Pemba	Chake Chake	22	446	20.3

6.1.2. Most impacted parts of the Chwaka Bay ecosystem

The findings of this study suggest that the fish species subjected to an unsustainably high exploitation are the emperor species *L. borbonicus* and *L. lentjan*. These findings are confirmed by outcomes from a participatory workshop conducted in September 2016 with 25 fishermen from Uroa, Marumbi and Chwaka village, representing all main gears in use (i.e. dragnet, handline, spear, trap). Fishermen generally perceived that catches of *L. lentjan* are declining (except for trap fisher, Fig. S.6.1. and Fig. S.6.2.). Emperors are relatively slower growing species and many of them such as *L. lentjan* attain a relatively large maximum size, increasing their vulnerability to overfishing. Emperors are one of the main target groups of the fishery of Chwaka Bay (Fig. S.6.5..). Despite *L. borbonicus* and *L. lentjan* three other emperor species are highly abundant, *Lethrinus variegatus*, *Lethrinus mahsena* and *Lethrinus harak*. The latter two have similar growth characteristics (Ebisawa and Ozawa, 2009; Grandcourt, 2002) as *L. lentjan* and thus may experience similar exploitation rates. Abesamis et al. (2014) characterized *L. lentjan* and *L. harak* as coral reef fish that show a relatively low to medium vulnerability to fishing. The authors used the fuzzy logic expert system developed by Cheung et al. (2005) to estimate the relative vulnerability of 145 coral reef fish belonging to 10 families using the life history characteristics and ecological traits of each species. One of the more vulnerable emperor species that have been identified by Abesamis et al. (2014) is the Spangled emperor *Lethrinus nebulosus*. This species has a relatively low abundance in the catch, which could be a symptom of strong exploitation highlighting the need to investigate the impacts of fishing on the overall emperor community.

Handline and trap fishermen also perceive a decline in catches of the snapper *L. fulviflamma*. Snappers are relatively similar to emperors in that many species are slow growing, relatively late maturing and attain similar large sizes (e.g. *Lutjanus argentimaculatus*). Nevertheless, the stock assessment conducted in this study characterizes this species as fully- but not overexploited. When fishermen were asked why they perceive a decline in this species, they argue that in previous times *L. fulviflamma* was more abundant within the shallow bay limits, whereas today they seemed to have moved towards deeper waters due to an increase in water temperatures. This would explain the difference in the perception of fishermen from the workshop and the conducted stock assessment. Furthermore, if this explanation holds true, it highlights the difficulty in attributing decreases in abundance to fishing. Nevertheless, *L. fulviflamma* is a species that attains a relatively small maximum length compared to other Lutjanidae caught by the fishery and was characterized as low to medium vulnerable by Abesamis et al. (2014). In contrast, *Lutjanus sebae*, *Lutjanus bohar*, *Lutjanus argentimaticulatus* and *Aprion virescens* are all species classified as highly vulnerable to fishing. Similar to the case of *L. nebulosus* the relatively low abundance in the catches could be attributed to strong exploitation. Chwaka Bay fishermen reported decreases in the catches of snappers already in 2002 (de la Torre-Castro and Rönnbäck, 2004). Thus, this group of fish, particularly the most vulnerable ones mentioned above require further research.

Findings from the ecosystem-based analysis in this study suggest that the biomass of herbivorous fish is relatively low and may limit the control on macroalgae cover. However, mean herbivorous fish densities estimated by Aller et al. (2014) across various sites around the islands indicate that Chwaka Bay hosts relatively high densities (Chapter II). Furthermore, the stock assessment analysis indicates that none of the analysed Scaridae species are showing signs of overfishing. Scaridae and Acanthuridae have been characterized by Abesamis et al. (2014) as reef families showing relatively low vulnerabilities. Thus, it is questionable if the fishery in Chwaka Bay has reduced the biomass of herbivorous fish to a level that top-down control is impaired and the system is in risk of excessive macroalgae proliferation. The only herbivore species that experiences exploitation rates that exceed biological reference points is *S. sutor*, the most abundant target species of the fishery. However, none of the fishermen from the participatory workshop perceive a decline in *S. sutor*'s catches (Fig. S.6.1. and Fig. S.6.2.). When asked why they think that this species is withstanding the current strong exploitation, they argue that part of the population is occurring further offshore and would come in periodic pulses into the bay. This suggests that *S. sutor* may spawn further offshore and might also be partly residing in deeper areas, resulting in a steady flow of recruits into the bay. This would be in line with observation from the Seychelles and Kenya of spawning aggregations of this species in waters deeper than 15 m (Bijoux et al., 2013). Such spatial refuge could increase *S. sutor*'s resilience to overfishing. Furthermore, Tyler at al. (2009) found evidence for such depth refuge on the west coast of Unguja, where species richness of fish communities was depleted in the shallow depth of fished sites versus protected sites, but no signs of depletion were found for deeper areas (< 10 m). Similar depth refuges were found for the spear fishery in Guam and the artisanal fishery in Fiji (Goetze, 2011; Lindfield et al., 2014). Seemingly "unsustainable high exploitation rates" of *S. sutor* have been reported for the south-coast of the Kenyan (Hicks and McClanahan, 2012; Kaunda-Arara et al., 2003) and the Seychelles fishery (Robinson et al., 2011), but no collapses of these fisheries have ever been reported. Despite a possible depth refuge of *S. sutor*, another explanation for its tolerance to strong fishing pressure could be its high resilience due to its life-history characteristics: (1) it has a very short life span (2.5 years found in this study), rapid growth and a high population turnover rate (Grandcourt, 2002) and (2) it is a dioecious spawner with many aggregation sites distributed over a relatively wide geographical area and a relatively long spawning period (Robinson et al., 2011, 2004).

Overall the fisheries resources of Chwaka Bay can be classified into 4 groups 1) High economic importance showing signs of overfishing (Lethrinidae); 2) High economic importance not showing signs of overfishing (Scaridae, Siganidae, *L. fulviflamma*); 3) High economic importance and risk of overfishing due to high vulnerability (Sphyraenidae); 4) Low economic importance, risk of overfishing due to high vulnerability (Serranidae, Mullidae, Labridae, Lutjanidae). All of the species assessed were highly abundant in the catches but are classified as low to medium vulnerable (Fig. 6.2.). Particularly, for Lutjanidae a closer look has to be taken at some of the more vulnerable species. Although Serranidae, Mullidae and Labridae might not constitute the largest part of the catch, these groups are characterized as highly vulnerable. Furthermore, fishermen specifically reported decreases in Serranidae catches (de la Torre-Castro and Rönnbäck, 2004), highlighting the need for further investigation.

When examining the catch composition for the whole fishery of Zanzibar (Fig. S.6.6.), it shows that the here identified potentially impacted families (Lethrinidae, Lutjanidae, Serranidae and Mullidae) are also the most caught families throughout the island. In addition, the large catch of elasmobranchs, should be worrying since this is a highly vulnerable group, as outlined above and thus requires further investigation.

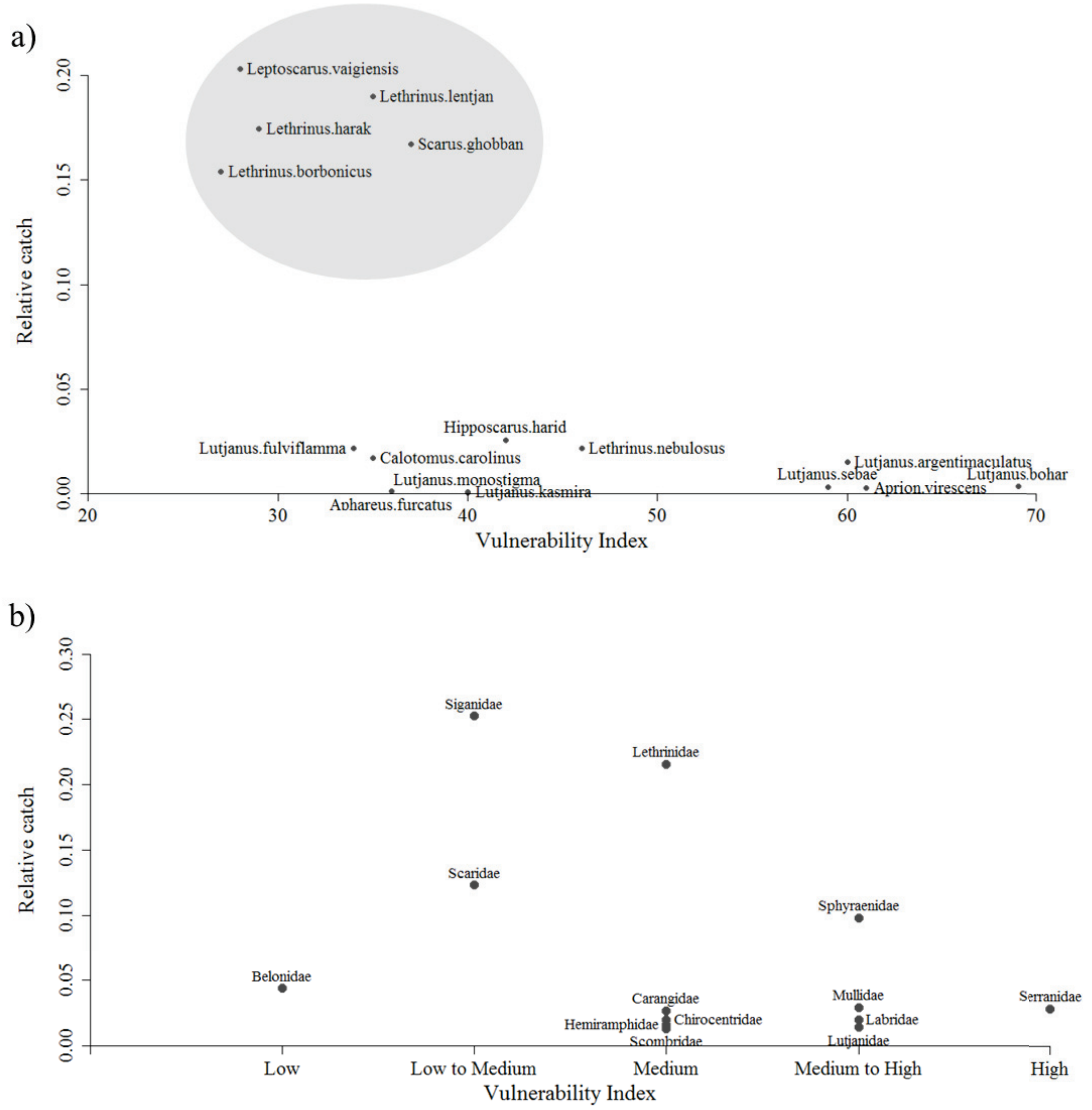

Fig. 6.2. Relative vulnerabilities of a) Some of the most abundant species of the families Lethrinidae, Lutjanidae and Scaridae (Shaded area highlights the position of the key species) and b) the most abundant families found in the catches of the Chwaka Bay fishery. Information of vulnerability was taken from Abesamis et al. (2014) and Vivekananda et al. (2009).

6.1.3. Identifying fishing methods with the highest impact

Dragnets and other pull seine nets such as beach seines are considered highly destructive and are subject of various management interventions throughout the WIO region (de la Torre-Castro and Lindström, 2010; McClanahan and Hicks, 2008; RGZ, 2010; McClanahan and Mangi, 2001; Cinner, 2010). The findings of this study confirm that dragnets together with traps and handlines are the most unselective gears in use, both in terms of species caught and in terms of the mean size of the catch. Dragnets also have the greatest impact on the ecosystem through a strong negative impact on a wide range of functional groups. Through heavy fishing of sea urchin predators (Labridae, *L. mahsena*, Balistidae, McClanahan, 1995) and pelagic fish, dragnets potentially reduce top-down control of these species. More importantly, dragnets together with handlines strongly target several of the highly impacted emperor species (Fig. S.6.9.). Overall, the catch of dragnets shows on average a medium vulnerability, when taking into account the 6 most abundant families (Fig. 6.2.). In addition, this gear is strongly competing with most of the other gears in use, but represents the least profitable fishing method in the bay (Table 6.2.). Interestingly, findings from the participatory workshop demonstrated that fishermen perceive dragnets as one of the most productive gears yielding the highest catches and correspondingly high profits (see Fig. S.6.3. and Fig. S.6.4.). This misperception is likely stemming from the amount of catch per boat, which is often much higher for dragnet boats. Together with the hope for the “big catch” (de la Torre-Castro and Lindström, 2010), this misperception is likely to drive more young fishermen into using this gear.

While our findings confirm the strong negative impacts of dragnets on the ecosystem and the fishing community, analysis also suggests significant negative impacts on the system through a large number of active trap fisher. Traps show a strong impact on the herbivorous fish community and induce the highest fishing mortality on 4 of the 6 analysed key species. This includes the overexploited *L. lentjan*. In contrast, with the strong focus on herbivorous fish, traps target many of the less vulnerable species (Table 6.2.). Although the herbivorous fish community does not seem to be overfished, further increases in the effort of this gear might impair its top-down control in the system. The use of small mesh-sized traps is becoming increasingly common (Pers. Comm.), which has resulted in a low mean size of the catch of traps. These findings are important, as to my knowledge very few studies within the WIO region highlight the potential harm of traps on the ecosystem.

Handlines, although of a relatively little impact on the overall system, particularly target emperor species and induce the highest mortality on the overexploited *L. borbonicus*. In contrast, spears are more selective, have a less severe impact on the ecosystem and have a comparatively low impact on the key species as well as on the emperor community (Table 6.2.). However, spears have been identified as strongly damaging coral reefs particularly through the divers trampling over the reef (Mangi and Roberts, 2006) and therefore are repeatedly suggested for management interventions (McClanahan and Mangi, 2001; Hicks and McClanahan, 2012).

Floatnets, gillnets and longlines are the most selective gears largely targeting pelagic fish as well as large carnivorous fish in the system. Through the capture of needlefish (Belonidae) and halfbeaks (Hemiramphidae), floatnets target less vulnerable fish groups and generally have a very low impact on the system (cf. Fig. 6.2. and Fig. S.6.13.). Similar to dragnets, but in contrast to the most profitable longlines, this gear generates very low profits per fisherman (Table 6.2.). Gillnets and longlines are both largely targeting highly vulnerable fish groups (cf. Fig. 6.2. and Fig. S.6.12.). The gillnets that are used within the study limits are largely targeting rays, which have been classified as among the most vulnerable target groups due to their K-selected life-history strategies and their high position in the food-web (Stevens et al., 2000). Longlines also target rays together with Carangidae and Serranidae, which show medium to high vulnerabilities (cf. Fig. 6.2. and Fig. S.6.11.). In contrast, the gillnet fishery at the north coast of Unguja (Nungwi) largely targets tunas and billfishes, which are caught 25 to 30 km offshore (Mildenberger, 2015). Many tuna species are fast growing with an early maturation and inhabit a wide geographical area, which makes them more resilient to overfishing (Birkeland, 2017). It is likely that the catch of gillnet and longline fisher from Chwaka Bay varies depending on the fishing location (distance to shore). However, on the basis of the catch from within the study area it is clear that longlines and gillnets concentrate on the most vulnerable target groups of the system.

Table 6.2. Overview of the different ecological and economic impacts of the different gears in use in Chwaka Bay. Critical negative impacts are highlighted in red.

	Traps	**Dragnets**	**Handlines**	**Spears**	**Floatnets**	**Gillnets**	**Longlines**	**Fence**
Vulnerability of key groups	Medium	Medium	Medium to	Medium	Low to	High	Medium to	Low to
Targeting Lethrinidae	Yes	Yes	Yes	Yes	No	No	Yes (few)	No
Ecosystem impact (MTI)	Very high	Very high	Medium	Medium	Low	Low	Low	Low
Competition with other	High	Very high	Medium	Medium	Low	Low	Low	Low
Size selectivity	Low	Low	Low	High	Low	NA	NA	NA
Species selectivity	Low	Low	Medium	Medium	High	High	Medium	High
Profitability	Medium	Low	Medium	Medium	Low	Medium	Very high	Low
Job provision	Medium	Very high	Low	Low	Very Low	Very low	Very low	Very low
Habitat destruction	Low	High	Low	High	NA	Medium	NA	NA
Potential for increase	No	No	Little	Little	Yes	Yes	Yes	No
Need for effort reduction?	Yes (slightly)	Yes	No	No	No	No	No	No
Need for monitoring?	Yes	Yes	Yes	Yes	No	No	No	No

6.2. Methodological approach and its limitations

Despite its common use, the applicability of length-frequency based stock assessment in long-lived, slow growing species is associated with relatively high uncertainty (Gulland and Rosenberg, 1992), because the relationship between age and length becomes more uncertain, when approaching L_∞. Since age reading has become increasingly feasible for tropical species, growth and mortality estimates of the key species, particularly *S. ghobban*, could be improved by length-at-age data. Furthermore, the single species models used here rely on estimates of natural mortality, why it is well known, that appropriate estimates of this parameter are difficult to obtain (Pascaual and Iribarne, 1993). Furthermore these methods assume a constant natural mortality rate over the entire exploited lifespan of the species, despite the fact that predation varies across different size-classes and may be much higher in small individuals (Gislason et al., 2010). The lack of incorporating statistical uncertainty of natural mortality in these single species models is clearly a caveat.

The difference between the stock assessment outcome and fishermen's perception on the status of *S. sutor* has raised the question whether catch length-frequency distributions of target species represent the entire stock. This question is crucial for small-scale, tropical fisheries, because they are almost always limited by depth and distance to shore. As a consequence stock assessment conducted on fisheries dependent data might result in an overly pessimistic view, if only the accessible fraction of the stock is being exploited heavily.

I chose the here studied species based on their importance in terms of abundance in the catches of the fishery. As seen above, all of those species have been classified as low to medium vulnerable. However, it might be that several of the more vulnerable target species (not further assessed here) may be overfished. I therefore would recommend for future studies to select a mixture of highly vulnerable, economically important and highly abundant target species as indicator species of the fishery. For doing so, I recommend combining analysis of preliminary catch composition data together with interviews of local fishermen about the most impacted species in the catches.

While we could obtain a relatively comprehensive picture for most of the gears in use, our predictions as well as calculations of profits for gillnets and longlines are still uncertain. This is due to the fact that we would only sample them if they were used within the study area. As was discussed above, the catch composition, the profitability etc. could change very much when taken into account their complete annual fishing trips.

EwE like any other modelling approach is based on a set of assumptions that constrain the utility of its use. Discussing all the caveats and limitations of the underlying model assumptions has been conducted elsewhere (e.g. Plaganyi and Butterworth, 2004). Nevertheless, it is important to note that due to the high amount of data required for the model, I had to obtain many of the input values from a) studies conducted before the model year, b) other study areas within the WIO region or c) other model systems. Particularly the use of diet information obtained from other years or sources for some of the model groups was suboptimal. *EwE* does not provide plasticity in predator-prey interactions, although the search and consumption of prey is often strongly related to relative prey abundance. Particular uncertainty in the Chwaka Bay model is associated with the diet of most of the species/families in the functional group *pelagic fish* and several species in the group *other carnivorous fish*, since information was taken from FishBase. This is especially relevant for the findings of Chapter V, since the results of the *Ecosim* analysis are sensitive to non-linear behaviour induced by predation and competition effects. In addition, the grouping of *other carnivorous fish* should be refined based on vulnerability of target species to fishing. This is needed for a more realistic simulation of alternative gear and effort regimes and their subsequent impacts on the ecosystem. For example, the grouping of low vulnerable species (e.g. *Lethrinus variegatus*) together with high vulnerable species (e.g. *Aetobatus narinari*) into the group *other carnivorous fish* might have led to life history and feeding characteristics that allow for a stronger increase in certain gear types.

The Chwaka Bay model is mainly based on information of macrobenthos and fish, lacking proper parameterization of the *meiobenthos*, *sessile benthos* and *zooplankton* compartments. Furthermore, no information of the bacterial community is available, despite the strong flow of the macrophyta compartments into the detritus pool. Although the model serves the purpose of this study well, it is advisable to parameterize the benthic invertebrate community of the system better. This is of particular importance, if the model was to be used to evaluate the consequences of invertebrate collection on the system. A list of potential improvements to the model is depicted in Table 6.3.

Ecopath has been criticised for not accounting for uncertainty in input parameters, model outcomes and food-web structure (Hill et al., 2007). While *Ecosim* includes a Monte-Carlos approach to fit parameter-combinations to biomass, catch or effort time series data, the lack of any time series for Chwaka Bay made it impossible to "tune" the model in order to see how well it can predict "reality". However, the use of the *Ecoranger* routine in Chapter IV allowed for the construction of alternative models, providing confidence to qualitative outcomes of the steady-state model. One of the most sensitive parameters is the vulnerability of the functional groups to predation. We set the vulnerabilities of functional groups according to the group`s trophic level. Although this is common practice in the absence of time series data (Cheung, 2001; Cheung et al., 2002; Kluger et al., 2016; Bacalso et al., 2016), the sensitivity of simulation outcomes to different vulnerability settings could greatly enhance reliability of scenario outcomes. Furthermore, it should be noted that the steady-state assumption of *Ecopath* generally limits the ability to simulate changes that deviate substantially from the equilibrium conditions (Plagányi and Butterworth, 2004).

In conclusion, the constructed model cannot be used to set target reference levels or to quantitatively predict changes in biomasses and profits under different fishing scenarios. The Chwaka Bay food-web model rather provides indices for the overall ecosystem state and generates a comprehensive picture of the relative impacts of the different gears on functional groups, ecosystem properties and profits of the fishing community.

Table 6.3. List of recommended steps for the improvement of the Chwaka Bay Ecopath model.

Order of importance	**How to improve the model**	**Data collection required?**
1	Sensitivity analysis of selected vulnerabilities	No
2	Refining the functional compartment *other carnivorous fish* based on life history parameters of the species	No
3	Collecting missing information of the feeding behaviour of the main abundant species of *pelagic fish*, *other carnivorous fish* and *omnivorous fish*	Yes
4	Collecting information on the biomass of *meiobenthos*, *sessile benthos* and *zooplankton*	Yes
5	Estimating the catch and effort of invertebrate harvesting	Yes
6	Estimating the biomass and P/B of the benthic bacteria community	Yes

6.3. Recommendations for management and data collection in Chwaka Bay and Zanzibar

According to the UN Code of Conduct for Responsible Fisheries the "best scientific evidence available should be used to evaluate the state of the fisheries resources". In tropical artisanal fisheries this could be as little as anecdotal evidence. In such situations, when knowledge is low, associated risks to the ecosystem are relatively high, and a precautionary approach to management is required. However, the FAO guidelines for the Ecosystem Approach to Fisheries state that the precautionary approach has to be applied to "any undesirable outcome (ecological, social or economic)" (Garcia et al., 2003). As a consequence, fisheries managers often have to balance between ecological and socio-economic goals, especially where strong resource dependence increases the vulnerability to management. Management decisions are then influenced not only by the state of the resources or ecosystem but also by food security, employment and economic viability (Bene et al., 2007). Within this context, in the following section I will discuss the potential options of management for Chwaka Bay in particular and for Zanzibar in general.

As elaborated in Chapter I, due to the complex nature of small-scale and multispecies fisheries, input measures are the most feasible for managing such fisheries. Thus, I will focus here on mesh-size regulations, gear and effort control and spatio-temporal closures. The latter management option was not investigated within the scope of this thesis and is therefore discussed separately as part of the section "future research directions".

6.3.1. Mesh-size regulation

The findings of this study suggest that despite the large amount of undersized fish in the catches, exploitation rates are exceeding reference points only for sizes above length at first maturity. The only exception to this is *L. lentjan*, for which juveniles experience unsustainably high exploitation rates. This finding is particularly important given that the observation of large amounts of small fish in the catches is often reason enough to label small-scale fisheries as unsustainable (Kolding et al., 2014) and to call for mesh size regulations (Mangi and Roberts, 2006; McClanahan and Hicks, 2011). However, an increase in mesh size to sustainably harvest *L. lentjan* will likely result in the strong decline of catches of *L. borbonicus*, *L. fulviflamma* and *L. vaigiensis*. This represents the dilemma of multispecies fisheries: harvesting one species at optimum level likely leads to the under- or overexploitation of most other species. Furthermore, due to the nursery characteristics of the bay, the depth limitations of the fishery and the ontogenetic habitat shift of several of the target species, it is questionable if an increase in mesh size would yield enough fish to economically sustain the fishery. Smaller specimens are usually more abundant and productive and can sustain higher mortality rates (Law et al., 2012). Currently, the Chwaka Bay and Zanzibar's resources in general sustain a large number of fishermen (Kathib and Jiddawi, 2010). Concentrating this fishing effort (fishing mortality) on the more vulnerable larger specimens will likely worsen the situation. Furthermore, small-scale fishermen very often do not comply with mesh-size regulations and it has been observed that unregulated or weakly enforced small-scale fisheries naturally develop an overall fishing pattern that matches the productivity of individual stocks fairly well (Kolding and van Zwieten, 2011; Law et al., 2012).

Thus, in order to sustainably manage the emperor community (Particularly, *L. lentjan* and *L. borbonicus*) and other potentially overexploited species, fisheries managers are likely left with either a reduction in fishing effort, gear management or spatio-temporal closures.

6.3.2. Gear and effort management

The strong negative impact of dragnet fishing on the overexploited emperor species (Chapter II), the ecosystem, and the profits of fishermen (Chapter III) together with the wide disapproval of this gear by fisheries managers and fishermen themselves (McClanahan et al., 1997), make it highly suitable for regulations. However, this study has shown that due to the provision of jobs a high dependency of the fishing community on the dragnet fishery in Chwaka Bay exists. The high dependency is the reason why none of the management interventions aimed at banning this gear have been successful to date and this situation is likely to worsen due to population and tourism increase (Lange and Jiddawi, 2009). Dragnets are not only in use at the east coast of Unguja, but throughout Zanzibar (Kathib and Jiddawi, 2010). Despite, their use being officially prohibited, enforcement is only effective inside managed areas, of which most are no-take zones (e.g. Chumbe). Thus, management interventions aimed at prohibiting this fishery need to be planned appropriately.

Our findings suggest that indeed a complete ban of dragnets would be most beneficial for the Chwaka Bay ecosystem and the individual profits of the remaining fisher, largely because it translates into a large reduction of fishing effort and per capita increase in catches. Thus, while from an ecological perspective the elimination of dragnets would seem as the most beneficial approach, the social and economic perspectives are ambivalent. The elimination of dragnets improves the economic situation of the local fishermen at the expense of the dragnet fisher, who would lose their prime source of income.

One way to avoid the loss in livelihoods is to reallocate the dragnet fishermen to other gears. This has already been attempted, but was largely ineffective because of the low number of alternative gears that have been provided (Gustavson et al., 2014). A reallocation of dragnet fishermen to traps and handline boats, however, would result in a strong increase in boat use, likely resulting in spatial use conflicts and increased pollution. Furthermore, our findings question the potential of Chwaka Bay's resources to withstand a large increase in traps, handlines and spears. In fact, simulations suggest that under a complete reallocation of dragnet fisher proportional to the effort of the other gears, the overall fish biomass and individual profits would largely decrease. Such reductions could have severe impacts on the food security of Chwaka Bay's community, since fishermen already earn close to the poverty line and many are concerned about the decreases in catches. Furthermore, the already overexploited emperor species *L. lentjan* and *L. borbonicus* would be further impacted through the increase in trap and handline boats. It has to be noted that an enforcement of the dragnet ban from the bay (or from other sites of Zanzibar) will inevitably lead to an increase in the use of other gears. These will most probably include those gears that are used in the same shallow areas as dragnets. This is particularly likely in Chwaka Bay, since the protection from wave energy through the reef generates relatively stable year round catches within the bay proper (de la Torre-Castro et al., 2014). Effort of *inter alia* traps and handlines must therefore be expected to increase and therewith the fishing mortalities on key species. Given the findings of the simulations in such a situation, the amount of traps and handlines targeting emperor species must be regulated (through bait use and hook size) and the biomass of herbivorous fish monitored. These findings are crucial because gear exchange programs are promoted throughout the WIO (Gustavsson et al., 2014; Mwaipopo, 2008; Signa et al., 2008; Wells et al., 2010).

The fisheries that seem to have room for expansion are floatnets, longline and gillnets. As mentioned above the high potential for an increase of gillnets is probably biased by the grouping of its main target group rays into *other carnivorous fish*. The present findings need to be treated with caution, since longline and gillnets are only partly applied in the study area and it is uncertain what type of species they target further offshore. Despite being less profitable, floatnets are good alternatives to dragnets, because they are operated in a similar way and thus require a relatively high number of fishermen (avg. 5 fisher boat^{-1}) compared to longline fishing. More importantly, floatnets target less vulnerable species, which would reduce the pressure on the highly impacted emperors.

Our simulations indicate that none of the gears in use can be used for a full reallocation of dragnet fishermen without compromising profits and biomasses of target groups, because they are simply too many. These findings suggest that Chwaka Bay is experiencing an overcapitalization of its fishery and likely suffers from dwindling profits, particularly through the use of dragnets. In a wealth-based fisheries management approach one would try to mitigate overcapacity and the subsequent dissipation of rent by restricting fishing access through privatization and efficient fishing rights systems (Cunningham et al., 2009; Kolding and van Zwieten, 2011). However, it has to be acknowledged that dragnets provide a great capacity to absorb surplus labour and thereby largely contribute to the welfare of the bay and the coastal communities around the island (Bene et al., 2010). Hence, rather than eliminating this supporting pillar of the fishing community entirely, it is more advisable to only mitigate the use of dragnets by a long-term stepwise reduction in effort to reduce fishing pressure on the vulnerable target species of the fishery, to protect the seagrass meadows and corals of the bay and to increase individual profits from fishing. Effort reduction is not well-accepted in most small-scale fishing communities (McClanahan et al., 2012, McClanahan, 2008; Salayo et al., 2008), which is no surprise given the lack of alternative income possibilities. Cinner et al. (2008) showed that particularly fisher from poorer households are less likely to stop fishing and look for alternatives, when the fishery is declining, suggesting that many small-scale fisher are trapped in poverty and are not able to take risks to escape a situation of decreasing catch rates. Thus, effort reduction programs need to be combined with livelihood diversification programs. To support managers, more research should focus on the possibility to use the bay (Particular the mangrove area) for aquaculture activities. Another option is to induce community-based tourism activities, such as bird watching close to Chwaka Bay. Many fishermen take tourists to the reefs inside the bay for swimming and snorkelling activities. However the communication between fishermen and tourists is hampered by the lack of education, which limits their possibility to acquire customers. Mostly, these activities are initiated through hotel staff or managers. Given my personal experience, it is very likely that only a small amount of the payment reaches the local community, highlighting the need for educational programs. Education has been identified as one of the main approaches to escape social traps (Costanza, 1987).

The fishing community must be involved in setting such fishing effort reduction plans given the strong repercussions this management measure entails. However, the current marginalization of dragnet fisher by other fishermen and their illegal status prohibits any discussion about reasonable management interventions. Therefore, I believe that first dragnet fishermen have to be recognized as legitimate and vital components of the fishing community. Further, dragnet fishermen should form a separate committee which will be obliged to regulate and administer the overall number of operating dragnet boats and will be supervised by the fishing authorities. Forums need to be created, in which resulting plans and goals are discussed and evaluated by all stakeholders (incl. other gears). Such community-based co-management has been identified to often better achieve management goals (see e.g. Pomeroy, 1995; Chuenpagdee et al., 2006). Some of the advantages of such management approaches are an increased sense of ownership and responsibility as well as increased compliance and surveillance (Gutiérrez et al., 2011). In fact, community-based co-management has been successfully implemented in Menai Bay (Zanzibar, Tobey and Torell, 2006) and the Tanga region (Tanzania, Wells et al., 2007). The willingness of dragnet fishermen to regulate the access to their fishing activities can be greatly enhanced by illustrating the negative consequences of increased dragnet effort on the profits of individual dragnet fisher. Furthermore, the strong difference in mean catch per fisher between the different gears and maximum possible daily catch of a dragnet fisher will help in raising awareness of the relative low profitability and might help in reducing the flow towards the use of dragnets.

One must bear in mind, that a reduction in dragnet effort and the subsequent increases in profits will most likely attract more people to join the fishery through the use of any of the other gears. Thus, subsequent increases in particular trap and handline boats need to be monitored. In the long-run, a reduction in dragnet effort is only meaningful if it is accompanied by an overall effort regulation in the bay and plans for diversification of alternative livelihoods.

6.3.3. Data collection

One of the central problems of the data collection system on Zanzibar is that information is only available in a highly aggregated form, namely overall catch in kg and price per target family per district. This information is only of limited informative value; neither analytical nor holistic methods can be applied to such data. Very detailed information is in fact collected by the beach recorder: the catch in kg and price of each target family per gear, fishermen, boat and day. However, this information is stored in piles of data sheets and is not made electronically available. Clearly, the limited personnel and technical capacities are the root cause of this situation. The Frame Surveys, aimed at collecting detailed information about fisheries stock variables such as fishing effort is only conducted every four years and the last one was delayed due to lack of financing. In contrast in Tanzania Mainland, Frame Surveys are conducted biannually (Sobo, 2004). In addition, important equipment (esp. spring balances) is often lacking, leaving the beach recorder with no other choice than guessing the catch weight. While in parts of Kenya large time series data sets for catch and mean size of target species exist (see for e.g. McClanahan and Hicks, 2011), Zanzibar is lacking any temporal information on fisheries data at species level. This highlights the need for a better financial support of the DMFR for the collection of data that can be used to evaluate and manage Zanzibar's fisheries. It seems that at the present moment Zanzibar spends money in collecting data for the purpose of reporting it to the FAO. However, I believe that only little more effort is needed to generate information, which can also be used for a proper assessment and management of Zanzibar's inshore resources. For instance, through the regular catch monitoring of beach recorders, who record the number of boats and fishermen that went fishing at the day of data collection, information on fishing effort theoretically exists. Using raising factors, the total annual number of fishing trips per boat could easily be estimated. Annual catch together with annual effort could then be used in surplus production models to calculate the current yield in relation to MSY. If this would be reported separately for each district, management could be conducted at district level. This is particularly important as it has been shown that fishing pressure and fishing impacts on the island are not homogenous. I furthermore suggest that data collected from Uroa and Chwaka village are aggregated separately to monitor catch changes within the bay rather than at the central district level.

The complexity of the fishery, the chaotic situation at the landing sites and the limited manpower for data collection all make it very difficult to accurately sample and estimate information on overall catch of target families let alone precise information on fishing effort. Furthermore, catch information at family level likely masks the dynamics of important target species. I therefore propose to expand the data collection to include the easier collected length-frequency distribution data of some of the key target species. This data can be used to develop two indicators: 1) fishing mortality and 2) mean size of the species-specific catch. Fishing mortalities that exceed biological reference points in three consecutive years or alternatively a 30 % reduction in the mean size of the catch of one of the indicator species observed during three consecutive years should trigger management interventions. However, changes in indicators and management options first need to be presented and discussed with local fishermen. This is highly important because the overall aim of management should be the use of a precautionary approach to prevent a) undesirable ecosystem states as well as b) undesirable socio-economic conditions of the villagers. It has to be noted, however, that decreases in mean size of the catch are not necessarily an indication of overfishing, because it often is induced by fishery independent drivers such as climatic conditions or market demand (Kolding et al. 2014). Due to the lack of detailed data of species catch composition from other landing sites, I would recommend to use the key species identified within this study to use as indicator species: *L. lentjan* and *L. borbonicus* (representative for *L. harak*, *L. mahsena*, *L. rubrioperculatus* and *L. variegatus*), *Cheilio inermis* (representative for the family Labridae). Furthermore, it is necessary to identify representative species of the dominant and vulnerable families Serranidae, Mullidae and Sphyraenidae. However, the final identification of key species to be monitored needs to be done together with local fishermen in a participatory workshop. Taking Chwaka Bay as an example, the collection of length-frequency data could be conducted by a team of two local fishermen on 5 to 7 days per week at Chwaka and Uroa landing sites. The fishermen would have to be given a compensation payment of 10 000 TZS (4.5 US Dollar) per day. Labour costs will then amount to about 1.7 million TZS per year, which would have to be either financed through taxes on annual catches of the fishery (about 0.2 %) or alternatively through a fixed fee of 700 TZS per month and fishing boat. While the length-frequency data provide biological indicators to represent stock status, CPUE could be used as an indicator to assess the status of the fisheries profitability. This is an indicator that is much easier understood by fishermen and more importantly it is an indicator that is of much relevance to the fishing community. Again, target levels of CPUE and thresholds that should trigger management interventions need to be set during a participatory workshop with the fishermen and the fishing authorities. A detailed scheme about the different steps involved for monitoring, analysis and management of Chwaka Bay's resources is depicted in Fig. 6.3..

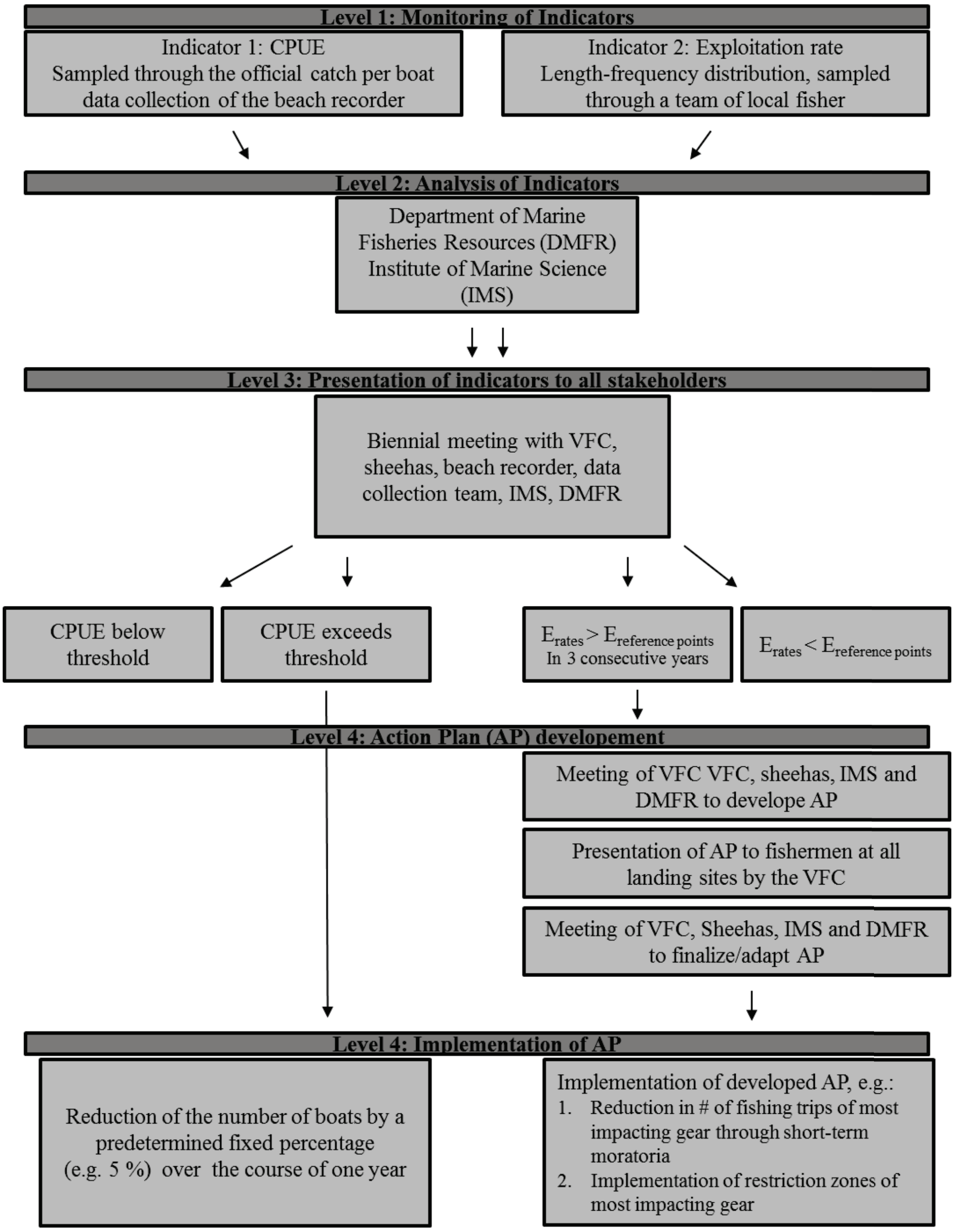

Fig. 6.3. Potential monitoring and management scheme for Chwaka Bay based on two potential indicators: CPUE, and exploitation rates obtained through a community-based sampling of catch length-frequencies of predetermined indicator species.

6.4. Research gaps and future research effort directions

6.4.1. Reallocation of Chwaka Bay's fishing effort offshore

One potential management option is to expand the radius of the fishery beyond the bay vicinity to fish in deeper waters. However, traps, dragnets and spears are generally limited to shallower depths, because traps need to be placed at the sea bottom and fishermen have to dive down to set the net or to hunt fish. Therefore, an expansion of the fishery would require a provision of appropriate boats and gears that can be used to fish in deeper water (e.g. gillnets and longlines). Projects aiming at enabling artisanal fishermen to fish further offshore have been conducted in different parts throughout the WIO region and were also part of the gear exchange program in Chwaka Bay (Gustavsson et al., 2014; Wells et al., 2010; Signa et al., 2008; Mwaipopo, 2008). However, the question is why so few fishermen in Chwaka Bay use gillnets and longlines and go offshore to fish, while this type of fishery is thriving in the North and the South of Unguja (Mildenberger, 2015). Dragnet fishermen argue this is due to the lack of financial capacity and access to this type of fishery (Wallner-Hahn et al., 2014). High costs associated together with the absence of financial means is also thought to be one of the reasons why gillnet fishermen are few in several fishing communities in the south of Kenya (Mangi et al., 2007). However, many fishermen in Marumbi received financial loans from foreign donors living in the village to buy fibre boats (Gustavsson et al., 2014). Furthermore, Chwaka village is one of the major markets on the island and as such many middlemen do business in the area. Hence, theoretically the potential for financial investment is present. Ultimately, the potential and success of a fishery depends on the relationship between costs and benefits. This includes considerations of the risks associated with fishing offshore at the east coast of Unguja. When winds and currents turn unfavourable or engines break, fishermen can be easily pushed towards the open ocean, contrasting to fishing within the Zanzibar Channel, where fishermen will be either pushed towards Tanzania mainland, Pemba or back to Unguja. Furthermore, on the East Coast fish might not concentrate on known sites, but rather be more scattered, while the channel could provide certain conditions for pelagic fish to aggregate and provide distinct, productive fishing grounds. While, all of this remains speculative, it points to the need to assess the potential for offshore fishing on the east coast of Zanzibar, prior to gear exchange programs aimed at reallocating fishermen to offshore fishing grounds.

6.4.2. Invertebrate harvesting

The exploitation of Chwaka Bay's resources is not restricted to finfish, but includes the extensive harvest of a wide range of invertebrates. Despite, the high dependency of the Chwaka Bay community on invertebrate collection for food security, the current data collection and management of the fishery focuses largely on finfish species and activities that use gears for fishing. However, the findings of this study indicate that the Chwaka Bay ecosystem is strongly benthic driven and inhabits a high abundance of invertebrates, suggesting that the system can support an intensive invertebrate fishery. Studies from the bay indicate that many species of gastropods, bivalves and sea cucumbers are showing reduced abundance and have decreased in the catches of individual fisher (Chapter II). This highlights the need to evaluate the dimension of gleaning activities in the bay and its impacts on the invertebrate community. This study fails at accurately estimating the effort of invertebrate harvesting and subsequently biomasses of target species are likely underestimated. Thus, it is not possible to evaluate the relative impact of this fishery on target groups or the ecosystem. Further studies are required to provide quantitative estimates of effort as well as evaluate the status of key target groups. Since gleaning activities are highly selective, temporal or spatial closures of vulnerable species are comparatively feasible.

The two most abundant invertebrate species of the fishery are *Sepioteuthis lessoniana* and *Octopus cyanea*. The findings of this study indicate that handlines and spears put great pressure on these two key species. Whether or not the current fishing mortality is sustainable should be investigated, given their importance for the fishery. Assessing their status and their market importance would constitute an ideal topic for two master students.

6.4.3. Spatial-temporal closures

In fisheries settings, like in Chwaka Bay where alternative income possibilities are low, only some of the target species show an unsustainable exploitation and the fishing pattern is highly adapted to the productivity of the fisheries resources, permanent or temporal area closures might be chosen instead as effort reductions and mesh size regulations. Particularly, because in such settings significant fishing effort reductions needed to protect key species are unlikely to be achieved in a short time frame. And more importantly, one of the key aspects in area management is that well-managed sites have the potential to protect key target groups without compromising fishermen's jobs (Sale et al., 2014).

Spatial-temporal management closures are one of the central management interventions throughout the WIO region (McClanahan et al., 2006, 2005b; Rosendo et al., 2011; Wells et al., 2007). The first formally recognised MPAs were established in 1965 and nowadays the majority of WIO countries have several sites of their inshore areas set under protection (Rocliffe et al., 2014). In Tanzania mainland, Kenya and Mozambique several of these sites are successful locally managed areas (Rocliffe et al., 2014). While the effectiveness of protected areas in the WIO region is difficult to assess due to a general lack of monitoring activities and corresponding data, there are some indications that several of the MPA's established have led to increased abundance of target resources as well as community benefits (Wells et al., 2007). For instance, the locally managed area Tanga coastal area (Tanzania mainland) in 1999 showed a general increased fish and invertebrate density shortly after implementation of protection (Wells et al., 2010). Another locally managed area in the central Kenyan coast likewise yielded in increased fish abundance of about 200 % (Rocliffe et al., 2014). A study by McClanahan et al. (2001) suggests that the protected Mombasa Marine Park (Kenya) led to enhanced yields per recruit of emperor species in adjacent sites through spill over effects. Furthermore, their findings indicate that protected areas can increase the diversity and decrease the variability of the fishermen's catch. The steady increase in the implementation of locally managed areas within the WIO region indicates a strong positive perception of community benefits by resources users (Rocliffe et al., 2014).

Zanzibar has formally established 3 networks of conservation areas (McLean et al., 2012): Pemba Channel Conservation Area, Mnemba Island-Chwaka Bay Marine Conservation Area and Menai Bay Conservation Area. However, most parts including Chwaka Bay remain general use zones and only Menai Bay Conservation Area is locally managed (Rocliffe et al., 2014). The only adequately enforced conservation areas on Zanzibar are the privately managed Mnemba Island Conservation Area and the Chumbe Island Conservation Area (McLean et al., 2012). As presented in Chapter II, several studies show that the Chumbe protected site hosts greater diversity, richness, abundance and biomass of target species compared to other unmanaged sites on Zanzibar.

Although Chwaka Bay has no proper zoning plan yet, the conflict between trap and dragnet fishermen has led to the designation of one small community-based dragnet-free zone in front of Marumbi village around 2002 (Marumbi Protected Area, Gustavsson et al., 2014). Initially Marumbi got support from the DMFR. However, nowadays Marumbi fishermen enforce the ban by themselves, due to a lack of financial capacity from the DFMR and most likely due to a lack of political willingness. According to fishermen, catches inside Marumbi Protected Area have strongly increased after implementation and many advocate for an expansion of the area. Thus, it is no surprise that during the participatory workshop, I conducted in 2016, fishermen developed an implementation plan for a no take zone within the bay when asked for potential community-based management measures to protect Chwaka Bay's resources and ensure future yields. Given the management experience of the WIO region and the willingness of fishermen for a community-based managed area, a spatio-temporal solution to protect the overexploited emperor species in the bay seems to be a feasible and promising management intervention. Nevertheless, experience has also shown that spill over to the remaining fishing grounds often do not compensate for the loss in overall fishing area (Abesamis et al., 2006; Agardy et al., 2011; Batista et al., 2014). For instance, in the case of Mafia Island Marine Park (Tanzania) fishermen perceived that the increase in effort within the general use zone has led to reduced benefits (Kincaid et al., 2014). After the implementation of the Mombasa Marine Park many fishermen stopped fishing in the area and searched for other fishing grounds or other income possibilities (McClanahan and Mangi, 2001). These aspects need to be taken into account when planning a no-take zone in Chwaka Bay. Fishing effort concentrations outside the protected site, potential losses in CPUE and local depletion of fish stocks need to be estimated prior to implementation and above all monitored during enforcement. Particularly the loss in CPUE is of great importance, because it might lead to reduce compliance or increased food insecurity.

In conclusion, the spatial management of Chwaka Bay's fisheries seems to be a promising way forward, but requires further research. The data I collected in 2014 included detailed information about spatial catch and effort allocation of the different gears in use, which can provide the scientific basis for proper zoning plans of the bay and information of potential consequences for the fishing community and the ecosystem.

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Annexes 1 and 2

Annex I – Supplementary Information

Supplementary information for Chapter I

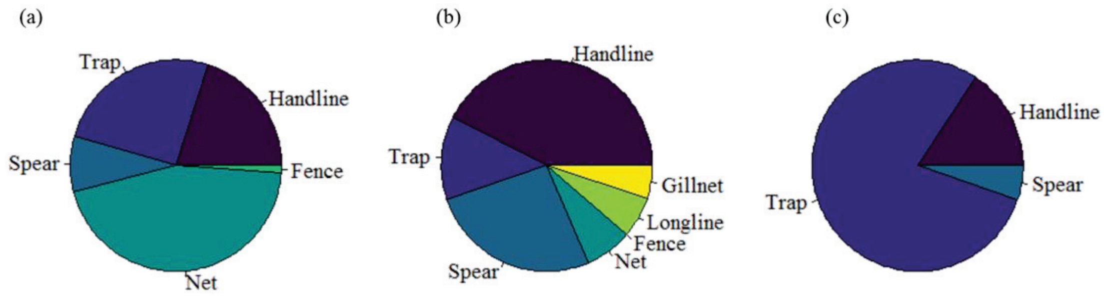

Fig. S.1.1. Relative dominance of gear types used in 2014 in a) Chwaka Bay, b) Uroa and c) Marumbi

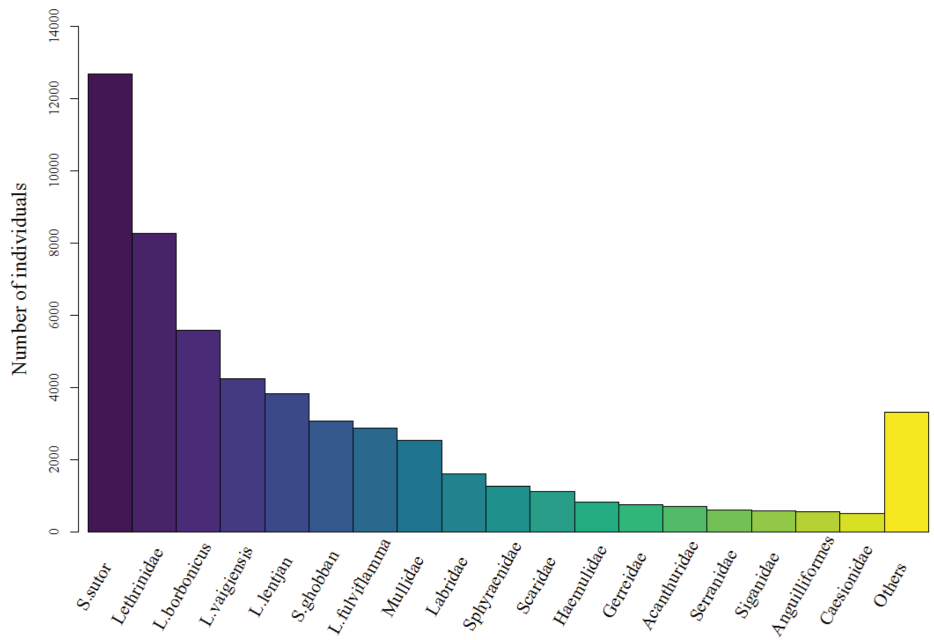

Fig. S.1.2. Number of individuals of the most important species/families found in the catches of Chwaka Bay in 2014.

Supplementary information for Chapter II

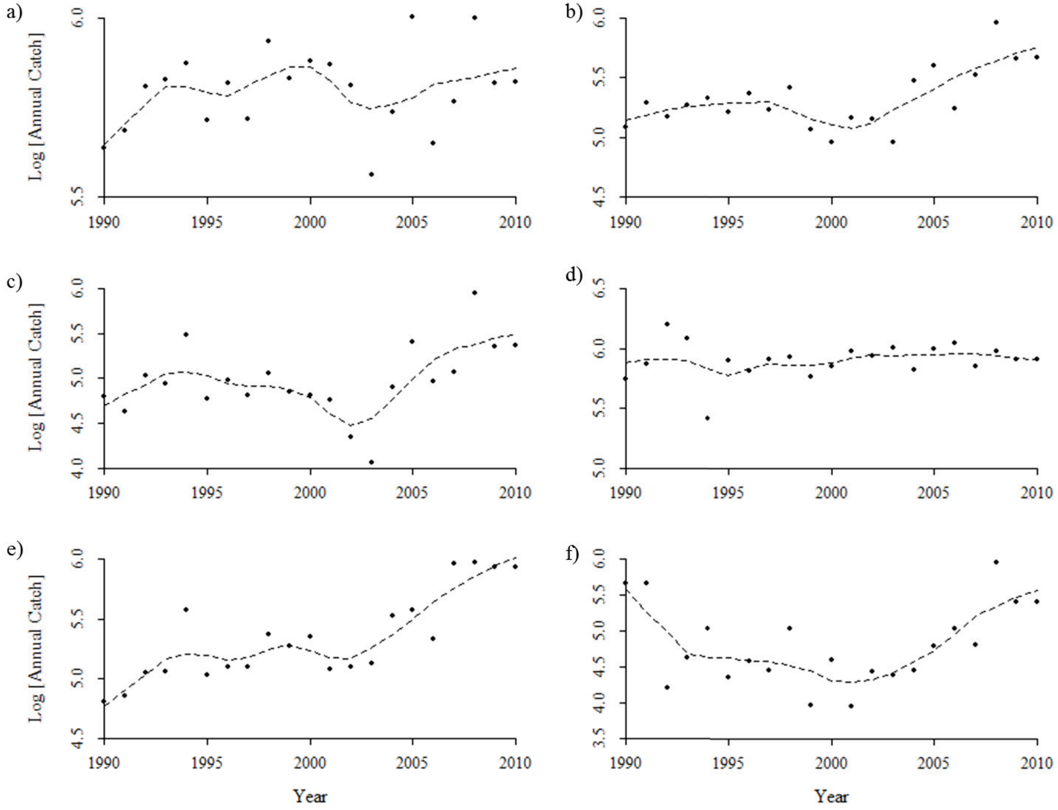

Fig. S.2.1. Annual catch of Unguja between 1990 and 2010 for a) Siganidae, b) Serranidae, c) Mullidae, d) Scombridae, e) Sphyraenidae, f) Lobsters . The dotted lines represent a LOESS smoothing of the data.

Table S.2.1. Data on Unguja's annual catch of 18 target groups as reported by the Department of Marine Fisheries Resources.

Unguja annual catches [kg]

	Siganidae	Scaridae	Lethrinidae	Serranidae	Mullidae	Lutjanidae	Mugilidae	Clupeioids	Sardines
1990	431766	231324	662696	122503	126002	25010	63273	953458	
1991	483105	260480	617195	192549	264388	87344	43021	1148216	
1992	644761	385835	655207	149278	248832	84073	106312	2064142	
1993	674002	362473	650071	186878	240802	136880	87464	1234281	
1994	745489	575847	946573	211177	383935	237927	305877	273294	
1995	518317	489894	865113	164367	229474	158212	59637	773414	
1996	655809	616441	947190	231626	321485	165172	94972	1007754	
1997	523852	495429	870648	169902	235009	163747	65172	778949	
1998	861590	605876	1713958	259348	205830	159070	114398	1441497	782280
1999	679498	219410	1876364	116852	145326	182822	69928	476828	1202206
2000	760389	321995	1385961	91852	189815	117381	63905	1248891	882393
2001	740518	317011	1111472	145128	1634465	323618	57634	1503644	489319
2002	649190	278898	1163801	143630	569398	444637	21852	2109192	615031
2003	365739	230711	656198	90805	236351	339944	11414	3316337	605367
2004	546119	405049	898484	296971	305562	256448	78944	2912093	1001503
2005	1004204	694689	748490	401962	348444	301684	257012	2701626	924894
2006	447902	312874	738361	172968	318514	422107	93577	2964767	687530
2007	582919	441849	1662428	333771	342362	293248	115744	2300085	519152
2008	995693	949455	1009712	911263	923106	926252	894857	1124863	991272
2009	656820	892605	1728112	453255	559866	423627	224782	1585541	760493
2010	663211	904155	1739531	462987	571115	430511	232716	1602948	777282

Table S.2.1. (continued)

Unguja annual catches [kg]									
	Mackerels	Carangidae	Tuna-like fish	King fish	Marlins & Sailfishes	Sphyraenidae	Sharks and Rays	Molluscs	Lobsters
1990	556927	407056	143644	191250	263031	64667	243876	333662	458250
1991	737432	132270	178027	194836	204445	71403	145443	485065	462828
1992	1601218	197226	203996	564625	112224	112973	599267	645857	15962
1993	1204652	207129	179241	258931	109601	115067	491731	366631	42095
1994	258297	409014	66319	336453	156181	373940	553379	530099	110257
1995	797857	176907	330078	310350	70589	108452	421901	629096	22592
1996	653544	221543	279049	220719	79471	126832	292352	596729	38908
1997	803392	182442	335613	315885	76124	125064	427436	634631	28128
1998	845975	265573	604949	394542	269837	235796	541745	552075	109691
1999	579906	269874	426690	257828	426282	190596	817772	555320	9376
2000	701769	177742	973987	791890	686070	225181	603300	439381	39511
2001	945201	195890	826417	598680	543717	119953	396577	410568	8923
2002	865719	308370	650683	551953	801253	125225	529926	416335	27783
2003	1024314	171202	485456	1074672	266438	134370	1016193	354267	24853
2004	667423	786864	556237	288923	425427	336017	1417488	567622	28567
2005	988589	408187	747563	537156	412451	378411	684359	928286	63076
2006	1106477	1265517	567619	578418	348601	216533	1098356	436430	107016
2007	704223	823664	593037	844875	462227	924132	825637	604422	65367
2008	955511	981838	1006431	969072	974688	951403	983562	1011602	893805
2009	808738	837212	1022648	858762	754152	862300	907006	748423	254664
2010	814752	850382	1026572	864750	758936	871157	919244	764130	256743

Table S.2.2. Data on Zanzibar's annual catch of 18 target groups from 1990 to 2012 as reported by the Department of Marine Fisheries Resources. Annual catch information for 2013 and 2014 was obtained from the FAO (*), except for Mackerels and Marlins & Sailfishes.

Zanzibar annual catches [kg]

	Siganidae	Scaridae	Lethrinidae	Serranidae	Mullidae	Lutjanidae	Mugilidae	Clupeioids	Sardines
1990	542991	360294	869504	184366	191362	79776	94757	1281132	
1991	610683	456004	808591	247000	333096	186516	86501	1394366	
1992	822436	659675	901776	227357	330054	196771	189346	2283395	
1993	855146	635108	881003	260179	330200	219238	148822	1433030	
1994	844990	830283	1176328	271431	471421	305918	351914	507474	
1995	662596	753402	1087273	221330	311356	261647	108482	1136530	
1996	795229	926723	1227358	309708	421551	268214	125075	1727122	
1997	670898	761704	1095575	229632	319658	269949	116784	1144832	
1998	1103760	795648	1932416	390982	388714	189256	317281	1711566	782280
1999	1370232	1023018	2065638	229384	282724	387945	147796	673547	1769848
2000	1181597	1006959	2489544	200238	282407	269120	150215	1424281	971513
2001	1109124	1056213	2177676	287580	1738723	548272	131902	3202188	571795
2002	1186393	827941	1957674	268006	707469	684074	94965	3772430	898975
2003	710070	801123	1411393	176748	363481	521355	107600	4881722	710989
2004	1100634	1082216	1697974	614521	628447	569481	159593	4055020	1132521
2005	1060479	1043148	1814021	522966	561443	513826	289615	5267033	956363
2006	1266456	1051814	1815830	563024	689505	507466	107445	3110044	1511218
2007	932141	1225951	2496438	574412	834166	578471	855179	2373685	972964
2008	1002568	1294875	2564879	684567	856782	623854	326849	2405482	1165826
2009	1012358	1377715	2667451	698125	864238	638491	346972	2445987	1172394
2010	1023315	1395073	2684047	714372	881214	664263	359130	2473310	1199302
2011	1573806	1471891	2523046	730402	1163580	733076	803463	2026231	1368813
2012	866802	1654177	2076979	1701731	2486184	2663191	1497115	1893175	798809
2013*	954000	1612000	2146000	1771000	2487000	2732000	1571000	1987000	936000
2014*	1090000	1663000	2085000	1801000	2626000	2778000	1715000	2047000	1076000

Table S.2.2. (continued)

Zanzibar annual catches [kg]

	Mackerels	Carangidae	Tuna-like fish	King fish	Marlins & Sailfishes	Sphyraenidae	Sharks and Rays	Molluscs	Lobsters
1990	645944	507367	229644	239826	293111	231039	306314	459783	464108
1991	798024	181118	224206	225239	241997	125923	216096	575981	475475
1992	1682080	276501	290402	618035	163720	157986	754390	817915	47602
1993	1281023	284543	247889	306666	159862	205508	623091	531453	58625
1994	329446	491356	138338	371380	186493	446369	713554	725599	129698
1995	880491	245856	464254	341286	117078	177445	528366	826769	33618
1996	723400	333498	390921	283817	130882	225297	451572	885399	55179
1997	888793	254158	472556	349588	125380	196824	536668	835071	41921
1998	959250	342646	834258	547214	491715	289438	614326	572710	125805
1999	945648	311665	575585	456414	823532	464193	898497	609794	216663
2000	1002202	469083	1307187	983574	973464	1083613	891176	1048707	305795
2001	1506247	511382	1216862	734176	710593	685025	641488	1037730	68431
2002	1357308	741704	1102030	685056	1013955	672029	870486	980496	30467
2003	1725709	471754	1069014	2013370	450059	649291	1245294	701427	64086
2004	1035635	1256424	1690499	535318	652124	812943	1841466	993715	80834
2005	1267818	990308	1442589	786243	757237	658815	1235709	1108404	64690
2006	1448826	1302697	2555946	860507	783333	542726	1442860	1233840	217873
2007	1164210	1184938	1427663	1041001	1002253	1253686	1338514	922233	463420
2008	1235486	1294521	1564875	1295357	1156238	1296458	1394587	1067643	395461
2009	1246870	1297462	1578462	1325684	1164258	1329485	1399846	1154892	393108
2010	1259142	1312117	1583990	1334284	1171018	1344171	1418358	1179033	396150
2011	1774722	1245610	1802199	1146437	1625355	1253879	2285029	1466817	911898
2012	1096707	1299075	2423075	1411964	766259	1427168	1708614	728795	1681659
2013*	-	1374000	2494000	1493000	-	1515000	1776000	817000	1695000
2014*	-	1208000	2422000	1626000	-	1256000	1614000	1461000	2190000

Supplementary information for Chapter IV

Table S.4.1. Sources of input parameters

	Group name	B (t m^{-2})	P/B ($year^{-1}$)	Q/B ($year^{-1}$)	EE	Diet matrix
1	*Siganus sutor*	Jones length-based cohort analysis (Rehren et al., submitted*a*)	*Z* from catch curve analysis (Rehren et al., submitted*a*)	Emperical formula Pauly and Palomares (1998)	Estimated by *Ecopath*	Lugendo et al., 2006, de la Torre-Castro et al., 2008
2	*Leptoscarus vaigiensis*	Jones length-based cohort analysis (Rehren et al., submitted*a*)	*Z* from catch curve analysis (Rehren et al., submitted*a*)	Emperical formula Pauly and Palomares (1998)	Estimated by *Ecopath*	de la Torre-Castro et al., 2008, Gullström et al., 2011
3	*Lethrinus lentjan*	Jones length-based cohort analysis (Rehren et al., submitted*a*)	*Z* from catch curve analysis (Rehren et al., submitted*a*)	Emperical formula Pauly and Palomares (1998)	Estimated by *Ecopath*	del Solar Escardo et al., 2015
4	*Lethrinus borbonicus*	Jones length-based cohort analysis (Rehren et al., submitted*a*)	*Z* from catch curve analysis (Rehren et al., submitted*a*)	Emperical formula Pauly and Palomares (1998)	Estimated by *Ecopath*	del Solar Escardo et al., 2015
5	*Lutjanus fulviflamma*	Jones length-based cohort analysis (Rehren et al., submitted*a*)	*Z* from catch curve analysis (Rehren et al., submitted*a*)	Emperical formula Pauly and Palomares (1998)	Estimated by *Ecopath*	Kimirei, 2012
6	*Scarus ghobban*	Jones length-based cohort analysis (Rehren et al., submitted*a*)	*Z* from catch curve analysis ()	Emperical formula Pauly and Palomares (1998)	Estimated by *Ecopath*	Plass-Johnson, 2011

Table S.4.1. (continued)

	Group name	B (t m^{-2})	P/B ($year^{-1}$)	Q/B ($year^{-1}$)	EE	Diet matrix
7	Other carnivorous fish	(Y*2) /(P/B), assuming E=0.5	*Z* from catch curve using *L*∞ and *K* from FishBase (Froese and Pauly, 2015); *Z* from Kaunda-Arara, 2003; Vivekananda et al., 2003; Chen et al., 2015; estimated using a fixed *P/Q* of 0.25	Emperical formula Pauly and Palomares (1998)	Estimated by *Ecopath*	del Solar Escardo et al., 2015; Gajdzik, 2014; de Troch et al., 1998; Lugendo et al., 2006; de la Torre-Castro et al., 2008; O'shea et al., 2013; Abdurahiman et al. 2010
8	Pelagic fish	(Y*2) /(P/B), assuming E=0.5	Bacalso and Wolff, 2014	Emperical formula Pauly and Palomares (1998)	Estimated by *Ecopath*	Lugendo et al., 2006; Gajdzik, 2014; Sivadas and Bhaskaran, 2009; Mablouké et al., 2013; Khodadadi et al., 2012; Sever et al., 2009;
9	Other herbivorous fish	(Y*2) /(P/B), assuming E=0.5	*Z* from catch curve using *L*∞ and *K* from fishbase (Froese and Pauly, 2015); Bacalso and Wolff, 2014	Emperical formula Pauly and Palomares (1998)	Estimated by *Ecopath*	de la Torre-Castro et al., 2008; Plass-Johnson, 2011; Dromard et al., 2014;
10	Zooplanktivorous fish	(Y*2) /(P/B), assuming E=0.5	Bacalso and Wolff, 2014; estimated using a fixed *P/Q* of 0.25	Emperical formula Pauly and Palomares (1998)	Estimated by *Ecopath*	de Troch et al., 1998; Froese and Pauly, 2015
11	Omnivorous fish	(Y*2) /(P/B), assuming E=0.5	Bacalso and Wolff, 2014; Chen et al., 2015; estimated using a fixed *P/Q* of 0.25	Emperical formula Pauly and Palomares (1998)	Estimated by *Ecopath*	Lugendo et al., 2006; de Troch et al., 1998; Froese and Pauly, 2015; Lin et al., 2007

Table S.4.1. (continued)

	Group name	B (t m^{-2})	P/B (year^{-1})	Q/B (year^{-1})	EE	Diet matrix
12	Octopus	(Y*2) /(P/B), assuming E=0.5	Hoenigs equation (1983), $\ln(Z)=1.23-0.832*\ln(t_{max})$. $t_{max}$from Guard and Mgaya, 2002	Estimated using a fixed *P/Q* of 0.25	Estimated by *Ecopath*	del Solar Escardo et al., 2015
13	Squids	(Y*2) /(P/B), assuming E=0.5	Bacalso and Wolff, 2014	Pauly et al., 1993	Estimated by *Ecopath*	Bacalso and Wolff, 2014
14	Crabs and lobsters	Commercial species: (Y*2) /(P/B), assuming E=0.5. Non-commercial species: Eklöf et al., 2005	Bacalso and Wolff, 2014	Bacalso and Wolff, 2014	Estimated by *Ecopath*	Chande and Mgaya, 2004; Dahdouh-Guebas et al., 1999; Jayamanne and Jinadasa, 1991; Kanciruk, 1980
15	Other crustaceans	Eklöf et al. 2005	Vega-Candejas and Arreguin-Sanchez, 2001	Vega-Candejas and Arreguin-Sanchez, 2001	Estimated by *Ecopath*	Zimmermann et al., 1979;Opitz, 1996; Primavera, 1996; Alarcon-Ortega et al., 2012;
16	Bivalves	Eklöf et al. 2005	Vega-Candejas and Arreguin-Sanchez, 2001	Vega-Cendejas and Arreguin-Sanchez, 2001	Estimated by *Ecopath*	Silina, 2011; Carpenter and Niem, 1998
17	Gastropods	Commercial species: (Y*2) /(P/B), assuming E=0.5. Non-commercial species: Eklöf et al., 2005, Lyimo et al., 2008	Bacalso and Wolff, 2014	Bacalso and Wolff, 2014	Estimated by *Ecopath*	Tan, 2008; Carpenter and Niem, 1998

Table S.4.1. (continued)

	Group name	B (t m^{-2})	P/B (year^{-1})	Q/B (year^{-1})	EE	Diet matrix
18	Sea cucumbers	Y*2) /(P/B), assuming E=0.5	Aliño et al., 1993	Estimated using a fixed *P/Q* of 0.25	Estimated by *Ecopath*	Aliño et al., 1993
19	Other echinoderms	Eklöf et al. 2005 and Eklöf et al. 2006	Bacalso and Wolff, 2014	Bacalso and Wolff, 2014	Estimated by *Ecopath*	Lyimo et al., 2011; Stöhr et al., 2012; Brogger et al., 2014
20	Annelids	Eklöf et al. 2005	Chavez et al., 1993	Chavez et al., 1993	Estimated by *Ecopath*	Fauchald and Jumars, 1979
21	Other meiobenthos	Estimated by *Ecopath*	Warwick and Price, 1979	Estimated using a fixed *P/Q* of 0.25	Guesstimate based on the relative biomass contribution of these groups values found in other similar studies (Bacalso and Wolff, 2014; Chen et al., 2008; Cruz-Escalona et al., 2007; Tsehaye and Nagelkerke, 2008; Vega-cendejas and Arreguı, 2001)	Opitz, 1996; Moens and Vincx, 1997
22	Sessile benthos	Estimated by *Ecopath*	Opitz, 1996	Opitz, 1996	Guesstimate based on the relative biomass contribution of these groups values found in other similar studies (Bacalso and Wolff, 2014; Chen et al., 2008; Cruz-Escalona et al., 2007; Tsehaye and Nagelkerke, 2008; Vega-cendejas and Arreguı, 2001)	Opitz, 1996; Bacalso and Wolff, 2014

Table S.4.1. (continued)

	Group name	B (t m^{-2})	P/B (year^{-1})	Q/B (year^{-1})	EE	Diet matrix
23	Zooplankton	Estimated by *Ecopath*	Opitz, 1996	Estimated by *Ecopath* using a fixed *P/Q* of 0.29	Guesstimate based on the relative biomass contribution of these groups values found in other similar studies (Bacalso and Wolff, 2014; Chen et al., 2008; Cruz-Escalona et al., 2007; Tsehaye and Nagelkerke, 2008; Vega-cendejas and Arreguı, 2001)	Silva et al., 1993
24	Corals	Estimated by *Ecopath*	Opitz, 1996	Bacalso and Wolff 2014	Guesstimate based on the relative biomass contribution of these groups values found in other similar studies (Bacalso and Wolff, 2014; Chen et al., 2008; Cruz-Escalona et al., 2007; Tsehaye and Nagelkerke, 2008; Vega-cendejas and Arreguı, 2001)	Opitz, 1996; Liu et al., 2011; Bacalso and Wolff, 2014
25	Phytoplankton	Kyewalyanga 2004	Kyewalyanga 2004	-	Estimated by *Ecopath*	-
25	Phytoplankton	Kyewalyanga 2004	Kyewalyanga 2004	-	Estimated by *Ecopath*	-
26	Macroalgae	Sjöö et al. 2011	Freire et al., 2008	-	Estimated by *Ecopath*	-
27	Seagrass	Gullström et al. 2006	Lyimo et al. 2006	-	Estimated by *Ecopath*	-
28	Detritus	Emperical formula from Pauly et al., 1993	-	-	-	-

Table S.4.2. Initial and final input parameters

#	Group name	B initial	B final	P/B initial	P/B final	Q/B initial	Q/B final
1	*S. sutor*	0.524	0.524	3.73	3.73	26.6	26.6
2	*L. vaigiensis*	0.174	0.174	2.09	2.09	20.39	20.39
3	*L. lentjan*	0.141	0.141	2.64	2.64	27.29	18
4	*L. borbonicus*	0.0898	0.0898	3.56	3.56	24.79	24.79
5	*L. fulviflamma*	0.128	0.128	2.12	2.12	27.84	17
6	*S. ghobban*	0.328	0.328	1.07	1.07	20	20
7	Other carnivorous	0.931	0.931	2.19	2.19	8.72	8.72
8	Pelagic fish	0.553	0.553	2.16	2.16	12.14	12.14
9	Other herbivorous	0.053	0.053	3.32	3.32	32.75	32.75
10	Zooplanktivorous	0.047	0.115	3.53	3.53	15.35	15.35
11	Omnivorous fish	0.02	0.114	2.87	2.87	10.53	10.53
12	Octopus	0.2	0.2	4	4	16	16
13	Squids	0.148	0.148	3.64	3.64	16.6	16.6
14	Crabs and lobsters	4.126	4.126	5.05	5.05	22	22
15	Other crustaceans	5.880	5.880	15.75	15.75	50.51	52.51
16	Bivalves	5.818	5.818	1.84	1.84	9.58	9.58
17	Gastropods	5.299	5.299	3.52	3.52	12.75	12.75
18	Other echinoderms	10.49	10.49	1.24	1.24	4.95	4.95
19	Sea cucumber	0.0379	0.0379	4.45	4.45	17.8	17.8
20	Annelids	11.429	11.429	4.5	4.5	22.5	22.5
21	Other meiobenthos	-	(6.114)	8.55	8.55	34.2	34.2
22	Sessile benthos	-	(21.598)	2	2	14.01	14.01
23	Zooplankton	-	(1.799)	40	40	-	(142.86)
24	Corals	-	(5.886)	2.3	2.3	7.15	7.15
25	Phytoplankton	17.19	17.19	82.24	82.24	-	-
26	Macroalgae	206	206	13.25	13.25	-	-
27	Seagrass	501	501	3.95	3.95	-	-

Table S.4.3. Sensitivity analysis done with Ecoranger (EwE Version 5). Shown are the effects of a 50% increase in input parameter (Depicted are effects larger than±0.20) on „missing“ parameters. Sensitivity is expressed as (estimated parameter – original parameter)/ original parameter.

	Input group no	1	2	3	4	5	6	7	8	9	10	11	12	13
	Input parameters	B P/B	B P/B	B P/B	B P/B	B P/B	B P/B	B P/B	B Q/B	B P/B	B P/B	B P/B	B P/B	B P/B
#	No. output parameter	EE	EE	EE	EE	EE	EE	EE	EE	EE	EE	EE	EE	EE
1	***S. sutor***	-0.333												
2	***L. vaigiensis***		-0.333											
3	***L. lentjan***			-0.321 /-0.333										
4	***L. borbonicus***				-0.333									
5	***L. fulviflamma***					-0.325 /-0.333								
6	***S. ghobban***						-0.333							
7	**Oth. carnivor. f.**							-0.314 /-0.333						
8	**Pelagic f.**								-0.285 /-0.333					
9	**Oth. herbivor. f.**									-0.333				
10	**Zooplanktivor. f.**								0.23		-0.333			
11	**Omnivor. f.**											-0.333		
12	**Octopus**												-0.333	
13	**Squids**													-0.333

Table S.4.3. (continued)

	Input group no	14	15	16	17	18	19	20	21	22	23	24	25	26	27
	Input parameters	B Q/B	B Q/B	B P/B	B Q/B	B P/B	B P/B	B Q/B	B EE	EE P/B	EE	EE P/B	B P/B	B P/B	B P/B
#	No. output parameter	EE	EE	EE	B	EE	EE	EE	B	B	B	B	Q/B	Q/B	Q/B
14	**Crabs a. lobsters**	-0.205													
15	**Oth. crustaceans**		-0.214												
16	**Bivalves**	0.265		-0.333											
17	**Gastropods**	0.307			-0.295										
18	**Othe. ecinoderms**	0.3				-0.333									
19	**Sea cucumber**						-0.333								
20	**Annelids**		0.211					-0.228							
21	**Oth. meiobenthos**							0.201	-0.405						
22	**Sessile benthos**							0.366		-0.333					
23	**Zooplankton**										-0.35				
24	**Corals**				0.229							-0.333			
25	**Phytoplankton**							0.206					-0.33		

References of supplementary material for Chapter IV

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Supplementary information for Chapter V

Considering that no reliable information on the number of operating boats in Chwaka Bay was available for any time period, we used the total number of landed boats per month that were recorded by the beach recorders (officially employed catch monitor personnel) in Chwaka village and Uroa village to infer a trend in fishing effort. Data available on the number of boats recorded included: July – December 2009 (Missing September), January – May 2011, January – December 2012 and January – June 2013. For trend analysis we first divided the number of boats recorded per month by the number of fishing days per month (25 days) and then calculated an average boat number for 2009, 2011, 2012, 2013. We assumed this to be an index for the number of operating boats per year in Chwaka and Uroa village. We used a logarithmic regression analysis on these 4 data points to describe the increase in boat use over time. This information was then used to estimate the effort multiplier for the scenarios (Table 5.1.).

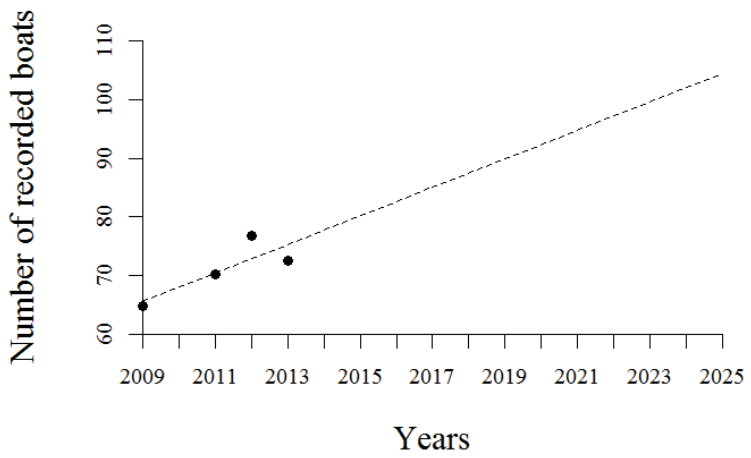

Fig. S.5.1. Extrapolation of fishing effort trend in Chwaka Bay using the average number of boats recorded per month in 2009, 2011, 2012 and 2013.

Supplementary information for Chapter VI

a) b)

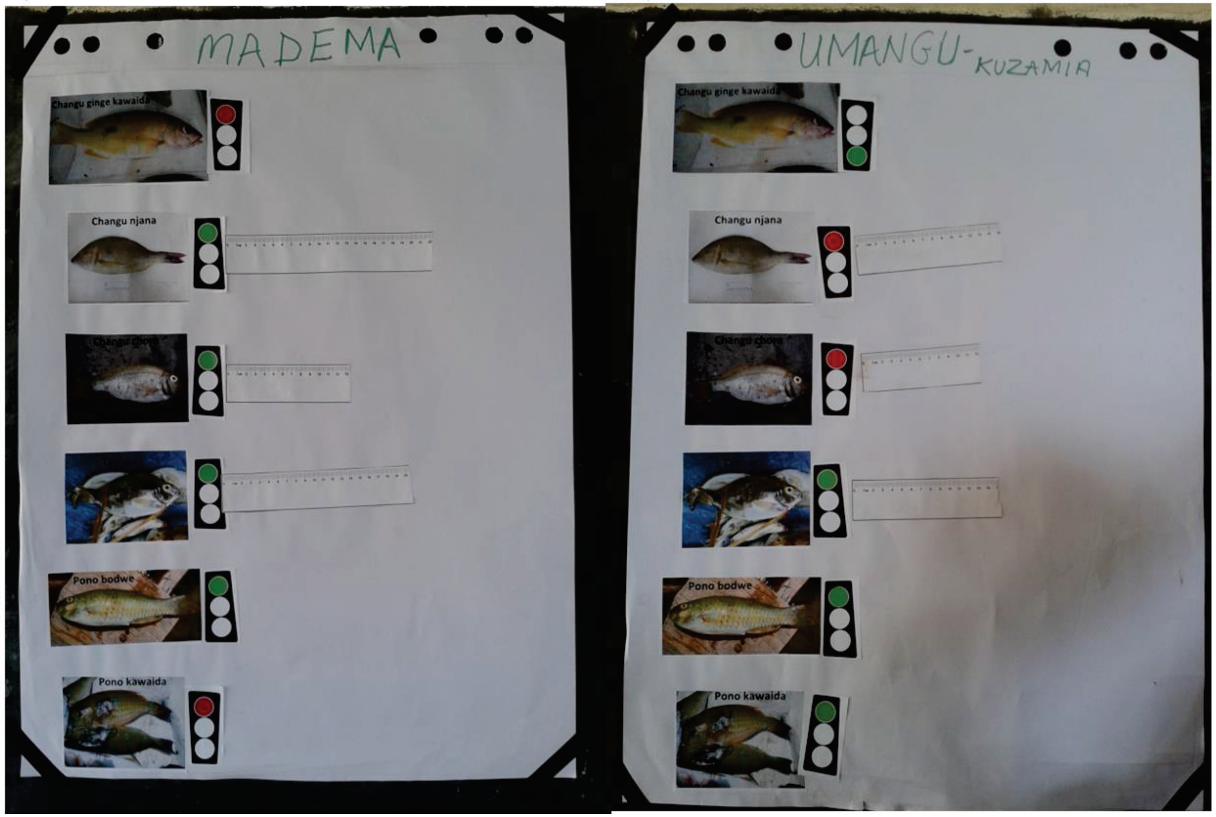

Fig. S.6.1. Results of the participatory workshop conducted with 25 fishermen in September 2016. Depicted is the perception of population decline for the six analysed key species (From top to bottom: L. fulviflamma, L. lentjan, L. borbonicus, S. sutor, L. vaigiensis, S. ghobban) of a) trap fisher and b) spear fisher.

c) d)

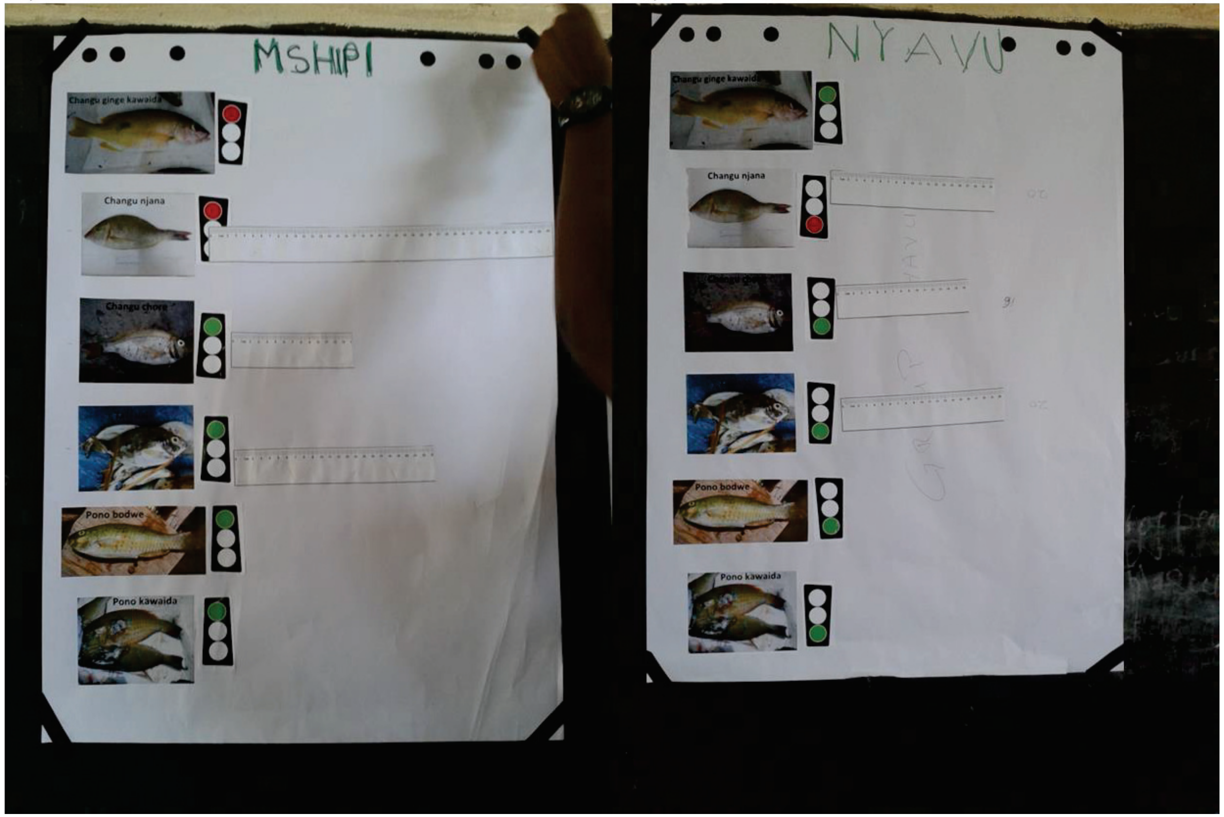

Fig. S.6.2. Results of the participatory workshop conducted with 25 fishermen in September 2016. Depicted is the perception of population decline for the six analysed key species (From top to bottom: L. fulviflamma, L. lentjan, L. borbonicus, S. sutor, L. vaigiensis, S. ghobban) of c) handline fisher and d) dragnet fisher.

a) b)

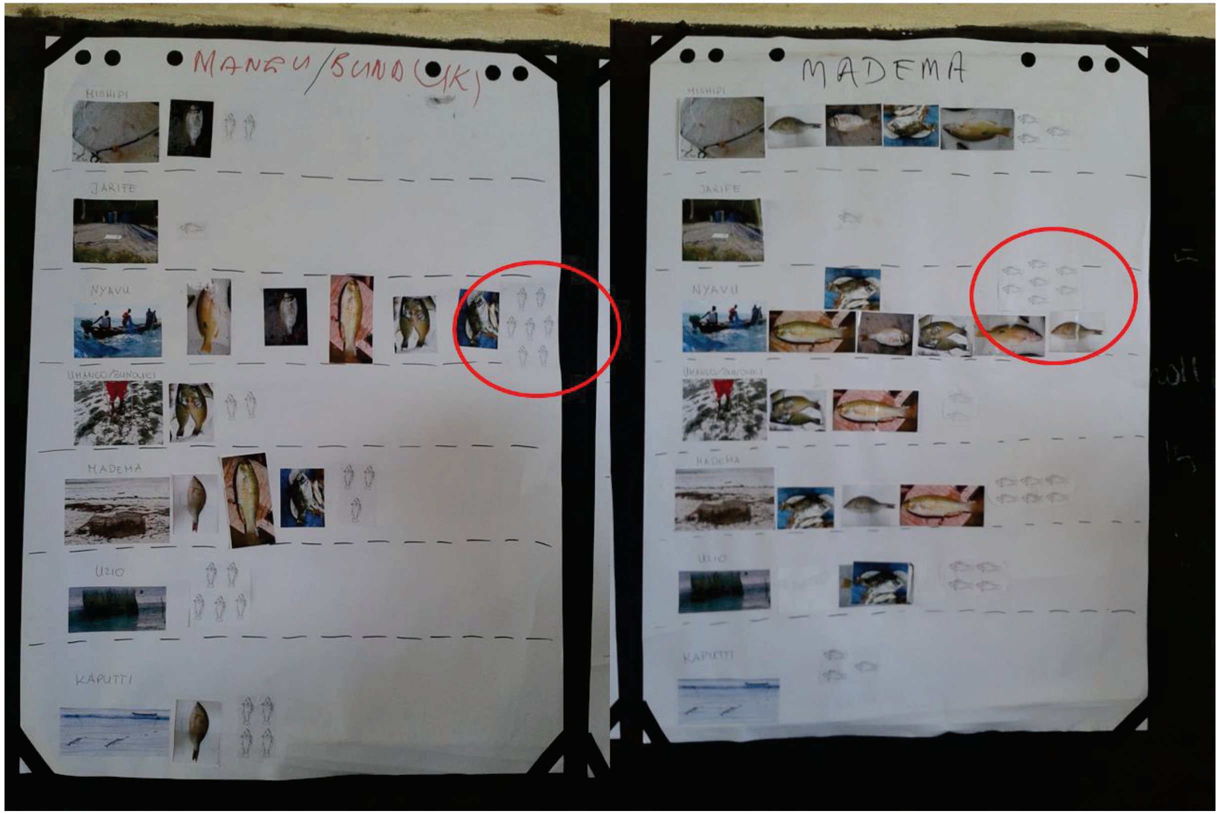

Fig.S.6.3. Results of the participatory workshop conducted with 25 fishermen in September 2016. Depicted is the perception of spear (a) and trap (b) fisher on the profitability of dragnet catch (marked with red circle). Fishermen were to rank the gears from producing low catch per fisher to high catch per fisher by using fish drawings ranging from 1 (lowest) to 7 (highest).

c) d)

Fig.S.6.4. Results of the participatory workshop conducted with 25 fishermen in September 2016. Depicted is the perception of dragnet (a) and handline (b) fisher on the profitability of dragnet catch (marked with red circle). Fishermen were to rank the gears from producing low catch per fisher to high catch per fisher by using fish drawings ranging from 1 (lowest) to 7 (highest).

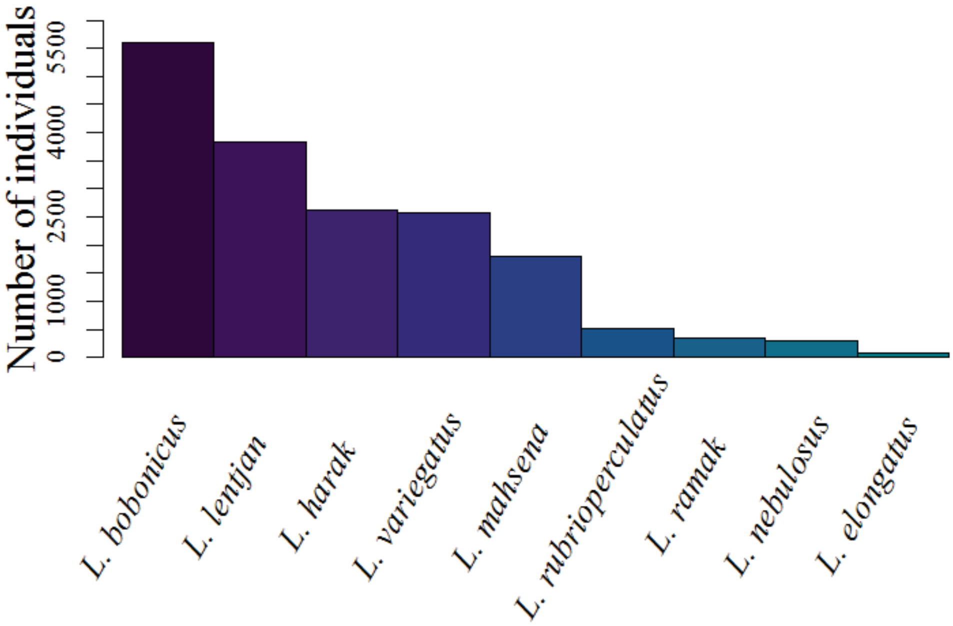

Fig. S.6.5. Relative importance of the different Lethrinidae species in the catches of the Chwaka Bay fishery. Depicted is the sampled number of individuals for each species obtained during the whole study period from the catches of dragnets, handlines, spears and traps.

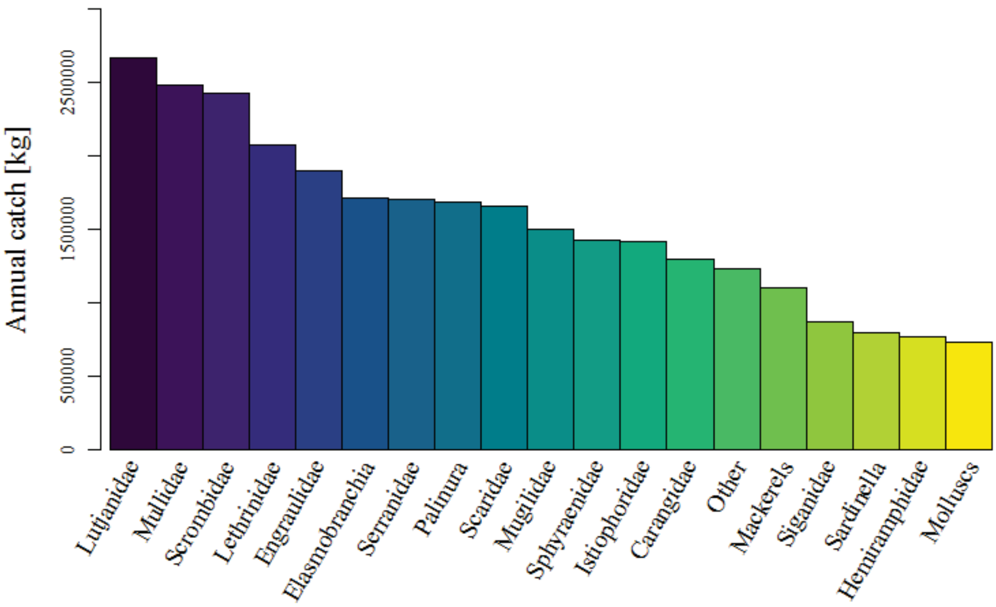

Fig. S.6.6. Total annual catch in kg of the 19 recorded target families/groups of Zanzibar reported in 2012. Data was obtained from the DMFR.

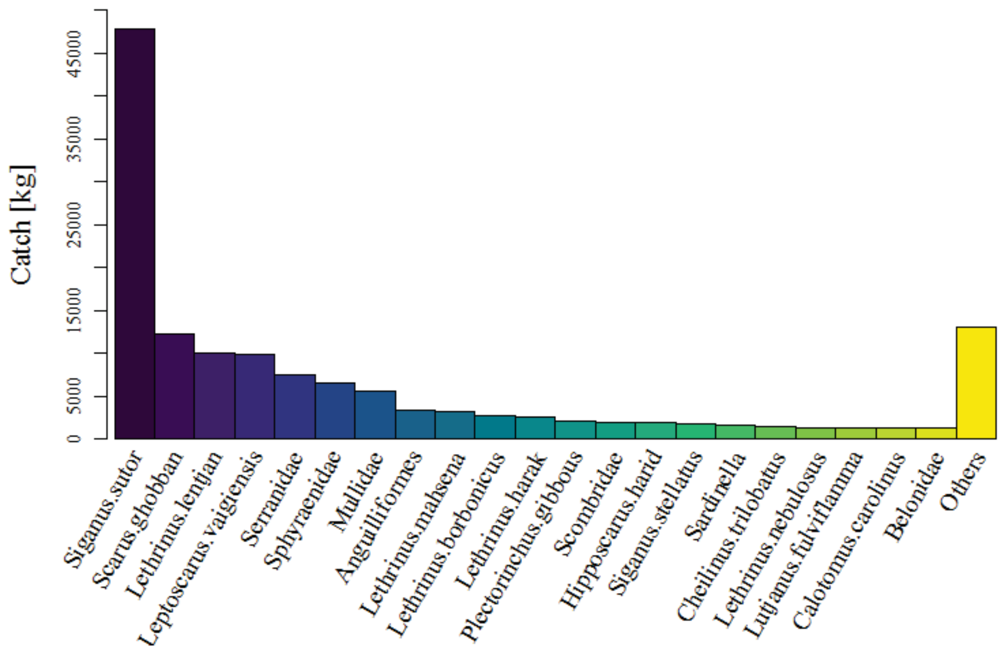

Fig. S.6.7. Catch in kg of the dominant target species/families (90 %) found in the sampled catches of trap fisher in 2014.

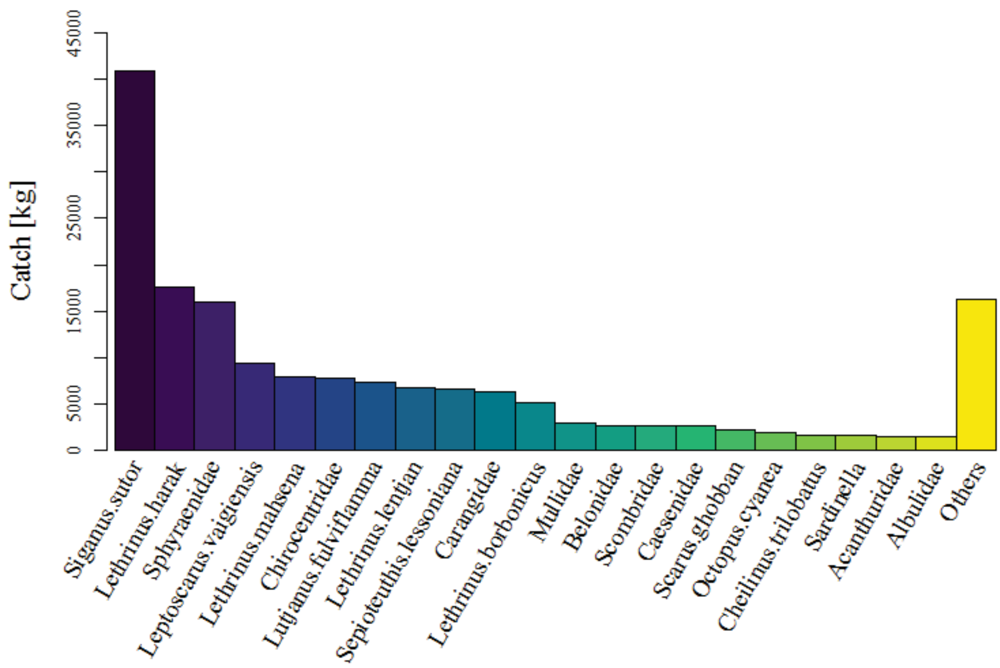

Fig. S.6.8. Catch in kg of the dominant target species/families (90 %) found in the sampled catches of dragnet fisher in 2014.

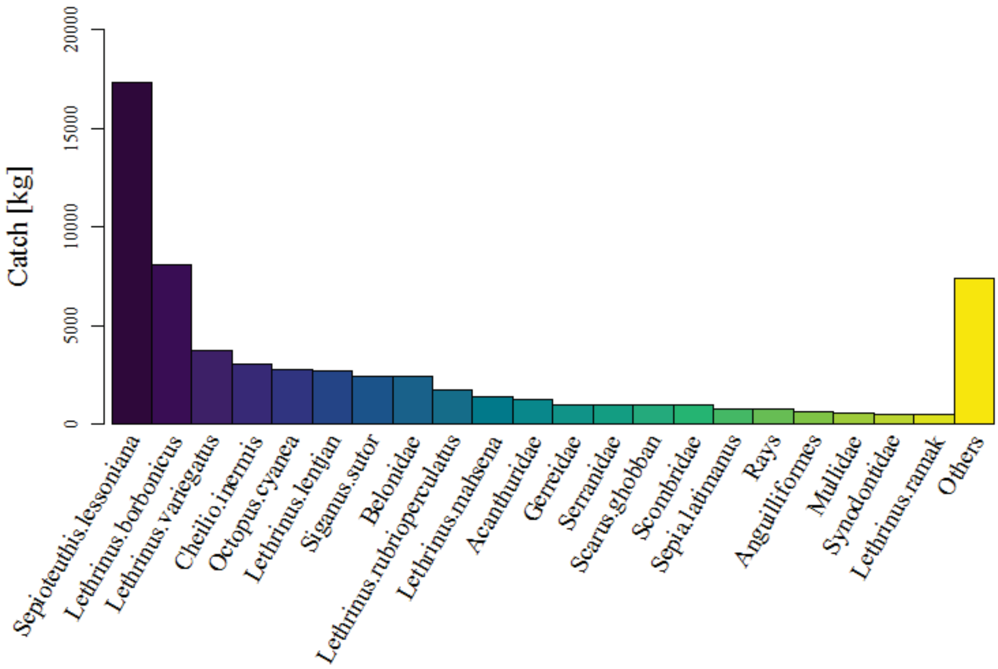

Fig. S.6.9. Catch in kg of the dominant target species/families (90 %) found in the sampled catches of handline fisher in 2014.

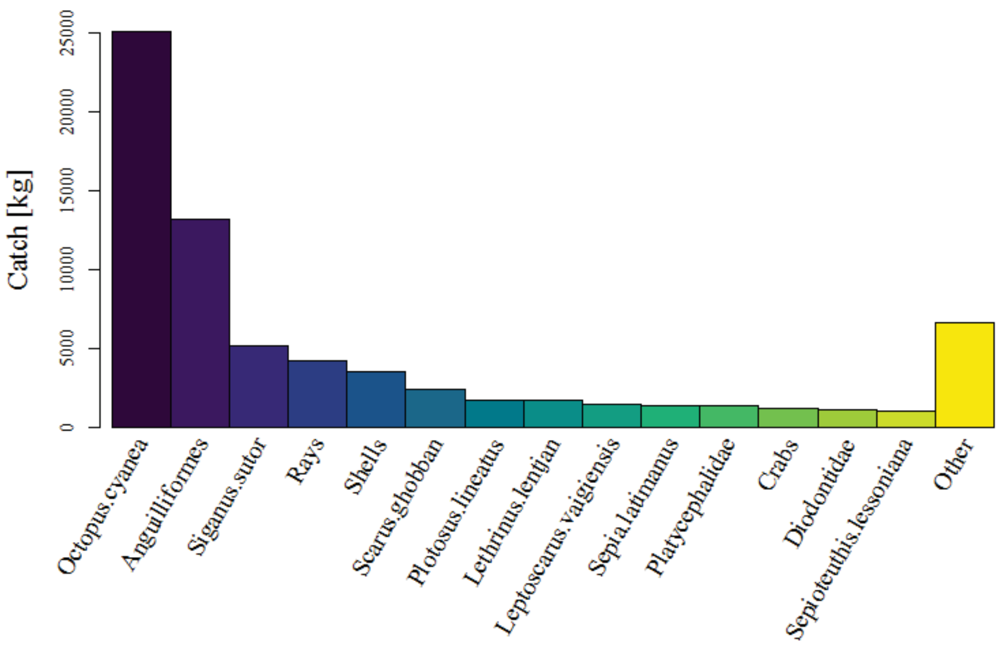

Fig. S.6.10. Catch in kg of the dominant target species/families (90 %) found in the sampled catches of spear fisher in 2014.

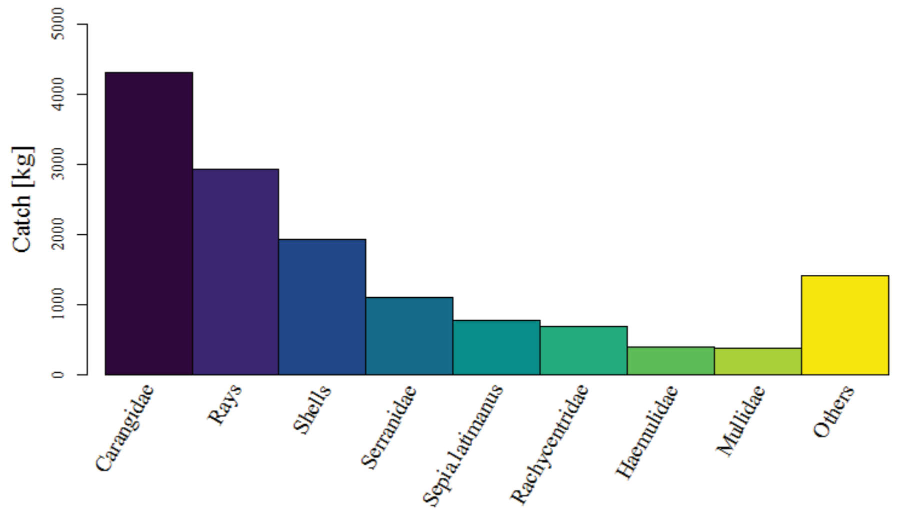

Fig. S.6.11. Catch in kg of the dominant target species/families (90 %) found in the sampled catches of longline fisher in 2014.

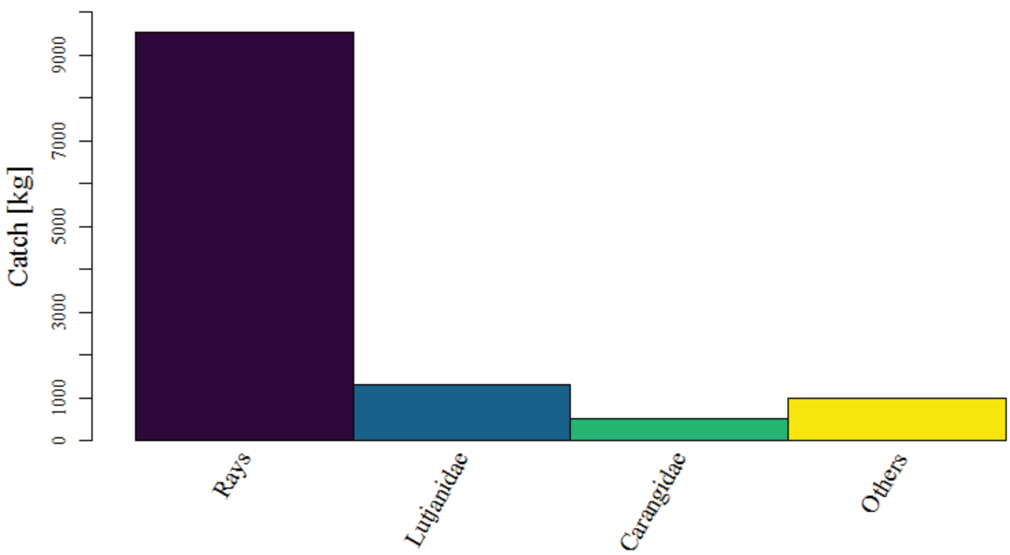

Fig. S.6.12. Catch in kg of the dominant target species/families (90 %) found in the sampled catches of gillnet fisher in 2014.

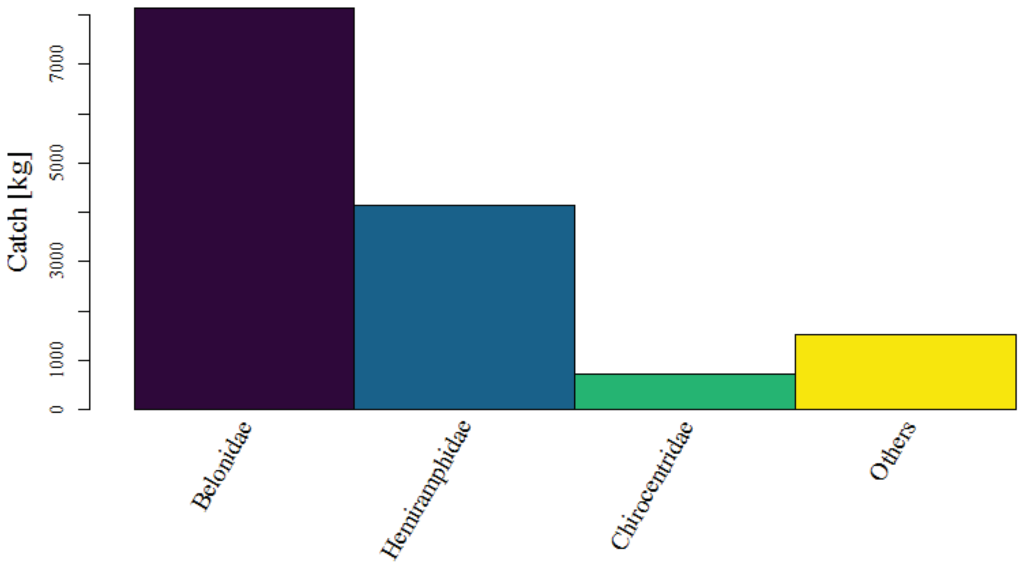

Fig. S.6.13. Catch in kg of the dominant target species/families (90 %) found in the sampled catches of floatnet fisher in 2014.

Annex II – Conference and workshops contributions

1. Rehren, J., Wolff, M., Jiddawi, N. **2017**. *Presentation:* Chwaka Bay (Zanzibar): How to sustain the fishery of this tropical bay system? ZMT Fisheries Workshop: Tropical Fisheries in a Changing World. Bremen, Germany. 2nd of February. http://www.zmt-bremen.de/Fisheries_Workshop.html
2. Rehren, J., Katikiro, R., Jiddawi, N. **2016**. Participatory Workshop with fishermen and village elders from Chwaka village, Uroa village and Marumbi village. 24th September.
3. Rehren, J., Wolff, M., Jiddawi, N. **2016**. *Presentation*: Chwaka Bay fisheries assessments. Final Symposium of the Leibniz Graduate School Sustainable Use of Tropical Aquatic Systems (SUTAS). Stone Town, Zanzibar, Tanzania. 19th of September.
4. Rehren, J., Wolff, M., Jiddawi, N. **2015**. *Presentation*: A Preliminary Evaluation of the Fishery of Chwaka Bay, Zanzibar Based on Stock- and Ecosystem Based Assessments. Wild Coast Sun, South Africa, Proceedings Books, Poster Abstracts, p.183. 28th of September.
5. Escardó, A.D.S., Wolff, M., Jiddawi, N.S., Rehren, J. **2015**. *Poster*: Diet composition and overlap of five commercial species of Chwaka Bay, Zanzibar, Tanzania. 9th WIOMSA Scientific Symposium. Wild Coast Sun, South Africa, Proceedings Books, Poster Abtracts, p.76.
6. Mildenberger, T.K., Wolff, M., Jiddawi, N., Rehren, J., Omar, O.H., Dewenter, J. **2015**. *Poster*: The artisanal tuna and billfish fishery on Zanzibar – description, quantification & valuation. Wild Coast Sun, South Africa, Proceedings Books, Poster Abtracts, p.129.
7. Rehren, J., Wolff, M. **2014**. *Presentation*: Modelling the multispecies fishery of Chwaka Bay, Zanzibar – basis for exploration of use and conservation scenarios. In: Steenbeek, J., Piroddi, C., Coll, M., Heymans, J. J., Villasante, S., Christensen, V. (eds.), Ecopath 30 Years Conference Proceedings: Extended Abstracts, pp. 58. Fisheries Centre Research Reports 22(3). Fisheries Centre, University of British Columbia [ISSN 1198-6727]. 237 p. 10th November.

Paper contributions

<u>Chapter II – State of the inshore fisheries in Zanzibar.</u>

Rehren, J., Wolff, M., Reuter, H., Jiddawi, N., **in prep.** Fisheries assessment of Chwaka Bay (Zanzibar) – following a holistic approach. (To be submitted to Afr. J. Mar. Sci.).

Contributions: The idea for this review was conceived by Jennifer Rehren. Jennifer Rehren revised the literature and is responsible for the data analysis. Jennifer Rehren wrote the manuscript, with improvements by all co-authors.

<u>Chapter III - Fisheries assessment of Chwaka Bay (Zanzibar) – following a holistic approach.</u>

Rehren, J., Wolff, M., Jiddawi, N., **(accepted with minor revisions)**. Fisheries assessment of Chwaka Bay (Zanzibar) – following a holistic approach. J. Appl. Ichthyol.

Contributions: The project idea was conceived by Jennifer Rehren and Matthias Wolff. Jennifer Rehren planned and conducted the data collection and is responsible for the data analysis. Jennifer Rehren wrote the manuscript, with improvements by all co-authors.

<u>Chapter IV - Holistic assessment of Chwaka Bay's multigear fishery – using a trophic modelling approach.</u>

Rehren, J., Wolff, M., Reuter, H., Jiddawi, N., **(accepted with moderate revisions)**. Holistic assessment of Chwaka Bay's multigear fishery – using a trophic modelling approach. J. Marine Syst.

Contributions: The project idea was conceived by Jennifer Rehren and Matthias Wolff. Jennifer Rehren constructed the *Ecopath* model and analysed the results with suggestions from Matthias Wolff. Jennifer Rehren wrote the manuscript, with improvements by all co-authors.

<u>Chapter V - Ecosystemic and economic effects of different fisheries management scenarios in Chwaka Bay (Zanzibar): a modelling study using EwE.</u>

Rehren, J., Wolff, M., Jiddawi, N., **in prep.** Ecosystemic and economic effects of different fisheries management scenarios in Chwaka Bay (Zanzibar): a modelling study using *EwE.* (To be submitted to Ecol. Modell.).

Contributions: Jennifer Rehren developed the simulation scenarios and analysed the results with suggestions from Matthias Wolff. Jennifer Rehren wrote the manuscript, with improvements by all co-authors.

Eidesstattliche Versicherung

Erklärung

Hiermit erkläre ich, dass ich die Doktorarbeit mit dem Titel:

Modelling the Multispecies Fishery of Chwaka Bay (Zanzibar) – Basis for Exploration of Use and Conservation Scenarios

selbstständig verfasst und geschrieben habe und außer den angegebenen Quellen keine weiteren Hilfsmittel verwendet habe.

Ebenfalls erkläre ich hiermit, dass es sich bei den von mir abgegebenen Arbeiten und drei identische Exemplare handelt.

23.05.2017

(Unterschrift) (Datum)